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Characterisation and environmental assessment of recyclable waste from recycling centres

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DTU Environment
Department of Environmental Engineering

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The synopsis part of this thesis is available as a pdf-file for download from the DTU research database ORBIT: http://www.orbit.dtu.dk.

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Preface

The work presented in this PhD thesis was carried out at the Department of Environmental Engineering of the Technical University of Denmark under the supervision of Professor Thomas F. Astrup, from October 2015 to December 2018. The work included a three months stay, from November 2017 to February 2018, at Fundació ENT (Spain) and collaborations with Danish waste operators.

The thesis is organized in two parts: the first part puts into context the findings of the PhD in an introductive review; the second part consists of the papers listed below. These will be referred to in the text by their paper number written with the Roman numerals **I-V**.

I Faraca, G., Boldrin, A., Astrup, T.F. (2018) Resource quality of wood waste: the importance of physical and chemical impurities in wood waste for recycling. *Submitted*. **[Paper I]**

II Faraca, G., Tonini, D., Astrup, T.F. (2019) Dynamic accounting of greenhouse gas emissions from cascading utilisation of wood waste. *Science of the Total Environment 651, Part 2, 2689-2700*. **[Paper II]**

III Faraca, G., Astrup, T.F. (2018) Plastic waste from recycling centres: relevance of waste characterisation data for modelling of recycling processes. *Submitted*. **[Paper III]**

IV Faraca, G., Martinez-Sanchez, V., Astrup, T.F. (2018) Environmental life cycle cost assessment: recycling of hard plastic waste collected at Danish recycling centres. *Resources, Conservation and Recycling (in press)*. **[Paper IV]**

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In addition, the following publications, not included in this thesis, were also concluded during this PhD study:

- Edjabou, M.E., Faraca, G., Boldrin, A., Astrup, T.F. (2018) Exploring the generation and composition of waste at household waste recycling centres in Denmark. *Submitted*.
- Faraca, G., Damgaard, A., Boldrin, A., Astrup, T., 2016. Life cycle assessment modelling considering impurities in recyclable materials. *Abstract from Life Cycle Assessment and Other Assessment Tools for Waste Management and Resource Optimization 2016, Cetraro (Italy), 6–11 June 2016*.
- Faraca, G., Boldrin, A., Damgaard, A., Astrup, T., 2017. Environmental assessment of presence of impurity materials and chemical pollutants in wood waste meant for recycling. *Abstract from SETAC Europe: 27th Annual Meeting – Environmental Quality through Transdisciplinary Collaboration, Brussels (Belgium), 7–11 May 2017.*
- Faraca, G., Boldrin, A., Damgaard, A., Astrup, T., 2017. Challenges to a circular economy – the presence of impurities in wood waste for recycling. *Proceedings Sardinia 2017, Sixteenth International Waste Management and Landfill Symposium. S. Margherita di Pula (Italy), 2–6 October 2017*
- Faraca G., Tonini, D., Astrup, T.F., 2018. Wood waste in a circular economy - dynamic accounting of greenhouse gas emissions from resource cascading. *Abstract from The 2nd Conference on Life Cycle Assessment of Waste 2018 – Symposium of Integrated Resource Management and Assessment, Borupgaard (Denmark), 18-22 June 2018*

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Summary

Recycling of waste has high priority in the European Union. The aim is to keep the functionality of resources within the anthroposphere, thereby reducing the pressure on the environment and increasing the security of supply. Despite an historical focus on energy recovery, Denmark is transitioning its waste management system towards increasing recycling of resources. However, while acknowledging the large achievements in the last decades, recycling of household waste still holds large room for improvements. Contamination is considered one of the main issues as it may act as technical, safety, and market barrier to recycling, ultimately affecting the quality of the recycled products, i.e. the ability to maintain the material properties comparable with virgin resources.

Some material fractions exhibit larger recycling difficulties than others. Wood waste, plastics waste and small combustible waste are waste fractions for which characterisation studies are needed to estimate contamination levels and for which alternative management solutions may enhance current recycling. Indeed, with increasing amounts of waste being recycled, there is a general interest to maximise the efficiency and effectiveness of recycling processes while ensuring clean and safe recycling loops. Similarly, it is necessary to ensure that recyclable resources do not enter waste streams not destined for recycling.

The goal of this PhD thesis was to assess the resource quality of wood waste, plastic waste and small combustible waste collected at recycling centres and link this to the recyclability of the selected waste fractions as well as the potential contribution to environmental savings. Recycling centres are manned collection points where the waste is sorted typically into 30-40 material fractions. In Denmark, recycling centres represent the only collection method for wood waste and small combustible waste, and receives around 40% of source-separated plastic waste.

Wood waste was sampled from three recycling centres and characterised according to product application, quality, and presence of material and chemical impurities. Overall, wood waste mainly comprised *Construction & Demolition* and *Furniture* applications, which showed the highest contamination by material impurities and chemical impurities, respectively. *Packaging* wood waste was the cleanest product application, although it contributed with a minor share to the overall wood waste composition. In particular, *Low quality* grade wood waste (i.e. wood waste treated for e.g. outdoor use and fibreboards) exhibited

dramatically higher content of chemical elements such as As, Cr, Pb and PAHs. Improving the management of wood waste may entail routing the *Low quality* grade to energy recovery through separate collection. Such alternative management system was tested by dynamic life cycle assessment (LCA). The results illustrated that global warming potential (GWP) savings could be increased by 5-58 times when recycling activities target only the upper qualities of wood waste. In this case, wood waste should preferably be recycled to floorboards or insulation boards, which can ensure substantial GWP savings due to substitution of long-lived or energy-intensive product.

Hard plastic waste, plastic film waste and PVC waste were sampled from three recycling centres and characterised in terms of product applications, quality, polymer, presence of material impurities and colour. The material composition appeared widely diversified across the three waste fractions: while plastic film waste mostly comprised *Non-food packaging* made of LDPE, PVC waste included only *Construction* applications. Conversely, hard plastic waste consisted of a wide number of applications and polymers, making it a very heterogeneous fraction to recycle. The *Low quality* applications (*Non-food packaging*, *Automotive*, *Construction* and *Other*) were characterised by a larger presence of non-plastic parts, multi-polymer products and coloured products than *High quality* applications (*Food packaging*). Consequently, the material losses in case of recycling *Low quality* plastic in a typical European mechanical recycling plant were 117% larger than recycling *High quality* plastic waste, as demonstrated by material flow analysis. As the effective removal of impurities influences the efficiency of recycling processes and the quality of the recyclates, three recycling scenarios were evaluated in terms of environmental and financial impacts. The results indicate that a mechanical recycling technology efficiently removing impurities can lead to large environmental savings and cost-effectiveness, as it produces recycled plastics characterised by high quality.

Small combustible waste was sampled from eight recycling centres and characterised with respect to material fraction, product application and, in the case of recyclable materials, other properties relevant when addressing the recyclability of the waste. Actual combustible materials constituted on average 38% of the waste, while 54% was identified as *Recyclable materials*, mainly in the form of plastics, textiles, paper and wood waste. If these Recyclable materials, currently incinerated as small combustible waste, were redirected to recycling processes, the national household recycling rate for glass, paper, cardboard, metals, plastics, wood, WEEE, textiles and garden waste (calculated after sorting losses) could be increased by 16%. Furthermore, the recycling of the Recyclable materials would enable save 27 kg CO2-eq/capita/year, representing an increase in avoided emissions of 30% compared to the current national savings from recycling the same recyclable fractions.

The role of quality was demonstrated to be crucial throughout the recycling chain of wood waste, plastic waste and small combustible waste, indicating that from a circular economy perspective “better recycling” may be preferable over “more recycling”.

Dansk sammenfatning

Genanvendelse af affald har høj prioritet i EU og har som formål at bevare ressourcers funktionalitet i antroposfæren og dermed reducere presset på miljøet samt højne forsyningssikkerheden. På trods af et historisk fokus på forbrænding, er man i Danmark begyndt at fokusere mere på genanvendelse af ressourcer. Samtidig med, at vi erkender den store fremgang i det sidste årti, er der stadig et stort potentiale for forbedring, når det kommer til genanvendelse af affald fra husholdninger. Her anses forurening for at være en af de største udfordringer, da forurening kan repræsentere en både teknisk, sikkerhedsmæssig og markedsrelateret barriere for genanvendelse som i sidste ende påvirker kvaliteten af det genanvendte produkt, det vil sige evnen til at bevare materialeegenskaber svarende til dem i virgine ressourcer.

Nogle materialefraktioner er sværere at genanvende end andre. Affaldstræ, plastikaffald og småt brændbart affald repræsenterer fraktioner hvor karakteriseringsstudier er nødvendige for at estimere niveauet af forurening og hvor alternativ affaldshåndtering kan forbedre genanvendelse. Med stigende mængder af affald der genanvendes, er der en general interesse i at maksimere virkningen og effektiviteten af genanvendelsesprocesser samtidig med at rene og ufarlige genanvendelseskredsløb sikres. Derudover er det vigtigt at genanvendelige ressourcer ikke ender i affaldsstrømme der ikke sendes til genanvendelse.

Formålet med denne PhD afhandling er, at vurdere ressourcekvaliteten af affaldstræ, plastikaffald og småt brændbart affald indsamlet via genbrugsstationer, samt koble kvaliteten af de udvalgte fraktioner til genanvendeligheden og potentielle miljømæssige gevinster. Genbrugsstationer er bemandede indsamlingssteder hvor affaldet bliver sorteret i 30-40 fraktioner. Genbrugsstationer er det eneste sted i Danmark hvor affaldstræ og småt brændbart affald indsamles og det er her omkring 40% af det kildesorterede plastikaffald indsamles.

Prøver af affaldstræ blev udtaget fra tre genbrugsstationer og karakteriseret i forhold til produktanvendelse, kvalitet samt tilstedeværelse af materielle og kemiske urenheder. Affaldstræ bestod hovedsageligt af bygge-relaterede produkter samt møbler, som indeholdte den største andel af henholdsvis materielle og kemiske urenheder. Emballageaffaldstræ var den reneste produktanvendelse, men repræsenterede kun en lille andel af den samlede affaldsmængde. Indholdet af kemiske stoffer, såsom As, Cr, Pb and PAH'er var markant højere i lavkvalitets træaffald (dvs. træaffald behandlet til udendørs brug og træfiberplader). Håndteringen af affaldstræ kan forbedres ved indsamle at lavkvalitetstræ separat og sende det til forbrænding. Sådanne alternative affaldshåndteringsmuligheder blev vurderet ved brug af dynamisk livscyklusvurdering (LCA). Resultaterne viste, at besparelserne i global opvarmningspotentialer (GWP) kan øges 5-58 gange, når genanvendelsesprocesserne kun håndterer højkvalitets træaffald. I sådanne tilfælde fortrækkes det, hvis træaffaldet genanvendes som gulvbrædder eller isolationspaneler, da dette vil resultere i markante GWP besparelser pga. erstatning af energiintensive produkter eller produkter med en lang levetid.

Prøver af hårdt plastikaffald, plastikfilmaffald og PVC affald blev udtaget fra tre genbrugsstationer og karakteriseret i forhold til produktanvendelse, kvalitet, polymertype, tilstedeværelse af materielle urenheder og farve. Materialesammensætningen varierede meget imellem de tre affaldsfraktioner: Hvor plastikfilm hovedsageligt bestod af ikke-madrelateret emballage lavet af LDPE, bestod PVC affaldet udelukkede af byggerelaterede produkter. Det hårde plastikaffald bestod til gengæld af en bred vifte af produktanvendelser og polymere og repræsenterede dermed en meget forskelligartet affaldsstrøm. Produktanvendelserne med lav kvalitet (ikke-madrelateret emballage, bildele, byggerelaterede produkter og andet) indeholdte større andele af dele der ikke var lavet af plastik, produkter lavet af flere polymere og farvede produkter, sammenlignet med produktanvendelse med høj kvalitet (mademballage). Via materialestrømsanalyse (MFA) blev det vist, at genanvendelse af lavkvalitetsanvendelserne i et typisk genanvendelsesanlæg derfor førte til materielle tab 117% større end genanvendelse af højkvalitetsanvendelserne. Eftersom virkningsfuld fjernelse af urenheder påvirker effektiviteten af genanvendelsesprocessen samt kvaliteten af det genanvendte materiale, blev tre genanvendelsesscenarier vurderet i forhold til økonomiske og miljømæssige påvirkninger. Resultaterne viste at mekanisk genanvendelse med en effektiv fjernelse af urenheder kan fører til store økonomiske og miljørelaterede besparelser, da der kan produceres genanvendt plastik af høj kvalitet.

Prøver af småt brændbart affald blev udtaget fra otte genbrugsstationer og blev karakteriseret i forhold til materialefraktion, produktanvendelse og, i tilfælde af genanvendelige materialer, andre egenskaber der blev anset som relevante i bestemmelsen af genanvendeligheden af affaldet. Egentlige brændbare materialer udgjorde i gennemsnit 38% af affaldet, mens 54% bestod af genanvendelige materialer, primært plastik, tekstiler, papir og træaffald. Hvis de genanvendelige materialer, der på nuværende tidspunkt forbrændes, i stedet blev sendt til genanvendelse, kan den nationale genanvendelsesrate (beregnet efter tab i forbindelse med sortering) stige med 16%. Derudover kan genanvendelse af de genanvendelige materialer føre til en besparelse på 27 kg CO2-eq/indbygger/år, hvilket svarer til en stigning på 30% sammenlignet med de nuværende nationale besparelser fra genanvendelse af de samme genanvendelige fraktioner.

Kvaliteten er essentiel hele vejen gennem genanvendelseskæden relateret til både affaldstræ, plastikaffald og småt brændbart affald, hvilket peger i retning af at ”bedre genanvendelse” er vigtigere end ”mere genanvendelse” i et cirkulært økonomi perspektiv.

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Abbreviations

CR-R	Collected for recycling rate
DK	Denmark
ELCC	Environmental life cycle costing
EoL-RR	End of life recycling rate
FTIR	Fourier-transformed infrared spectroscopy
FU	Functional unit
GC-MS	Gas chromatography mass spectrometry
GHG	Greenhouse gas
GWP	Global warming potential
HDPE	High-density polyethylene
ICP-OES	Inductively coupled plasma optical emission spectrometry
ICP-MS	Inductively coupled plasma mass spectrometry
iLUC	Indirect land use change
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LDPE	Low-density polyethylene
MAE	Microwave assisted extraction
MFA	Material flow analysis
PAH	Polycyclic aromatic hydrocarbons
PCB	Polychlorinated biphenyls
PCP	Pentachlorophenol
PET	Polyethylene terephthalate
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
RR	Recycling rate
SCW	Small combustibles waste
TC	Transfer coefficient
TH	Time horizon
UK	United Kingdom
WEEE	Waste electrical and electronic equipment

1 Introduction

1.1 Background

Municipal solid waste management systems have shown a change in direction in the last decades, transitioning gradually from permanent disposal treatments such as landfill and incineration in favour of recycling practices. In the European Union (EU), recycling is promoted through legislation such as the Waste Framework Directive and by establishing minimum recycling targets that EU Member States have to fulfil (EC, 2008; 2018c). Recycling and recovery processes are forestered in the quest toward a circular economy, enabling to keep the functionality of resources in the anthroposphere – and thereby reducing pressure on the environment and increasing the security of raw materials supply in the EU (EEA, 2016). Most European countries are therefore in the process of reconsidering their waste management systems. Denmark has committed to very ambitious recycling goals, despite being historically distinguished by a focus on incineration with energy recovery (The Danish Government, 2013).

The sustainability of recycling practices has been supported by the scientific community through the use of various metrics, such as resource efficiency indicators, process efficiencies, energy consumption, material flow analysis (MFA) and life cycle assessment (LCA) results, to name a few (e.g. Ardente and Mathieux, 2014; Shonfield, 2008; Turner et al., 2015). However, despite technological developmenst, recycling processes exhibit opportunities for further enhancement. For example, the European quantity of municipal solid waste sent to recycling was 45% of the total generated waste, in 2016 (EC, 2018b); however, large material losses are expected during recycling. Moreover, recycled products are not always able to replace virgin resources fully; for example, the contribution of recycled materials to primary materials demand in EU in 2016 was 12% (EC, 2018b). The substitutability of recycled products depends on their quality, i.e. the ability to maintain the material's properties (physical, mechanical, chemical and aesthetics) at a level that can compete with virgin resources (Ardente and Mathieux, 2012; Cimpan et al., 2015; Eriksen et al., 2018a). In addition, market and economic barriers may also constrain the utilisation of recycled products. However, only when these secondary products are able to replace primary resources can the benefits of recycling be fully realised (Michaud et al., 2010).

Contamination is acknowledged as one of the main issues preventing efficient and effective recycling of materials (Bilitewski, 2012; Lahl and Zeschmar-Lahl, 2013; Pivnenko et al., 2016b). Indeed, contaminants (impurities) may act as technical, safety and market barriers to recycling and the displacement of virgin resources (RDC Environment & Pira International, 2003). The presence of impurities is caused mainly by the large heterogeneity of waste, which requires thorough separation and restricts the effectiveness of property-based sorting. Characterisation studies may contribute to enhancing recycling by identifying aspects critical for recycling and setting the basis for management choices and technological development.

The heterogeneity of waste can be reduced by separately collecting similar material fractions (glass, paper, plastic, etc.) (Nilsson and Christensen, 2011). Separate collection may occur at the household premises (door-to-door, kerbside) or at collection points (recycling centres). Material fractions may also be separated by downstream material recovery facilities sorting mixed municipal waste, as initiated in Spain, Greece and the UK in addition to source separation schemes (Cimpan et al., 2016). Nevertheless, upstream measures are considered more effective than downstream sorting with respect to decreasing contamination of the waste (Villanueva and Eder, 2014).

Recycling centres are a popular collection method in Scandinavian countries, owing to a generally low population density and the abundance of single-family houses (Krook and Eklund, 2010a). In Sweden, one-third of household waste is collected at recycling centres (Krook and Eklund, 2010b), while in Denmark percentages may be up to 50% (Bisinella et al., 2017b; Miljøstyrelsen 2017a; 2018); in the rest of Europe recycling centres account for between 10 and 15% of waste collection (Villanueva and Eder, 2014). Recycling centres in Denmark are expected to play a key role toward enhanced recycling as the purity of the collected material fractions is expected to be high (The Danish Government, 2013). Typically, the waste is sorted into 30-40 fractions, and the presence of trained staff ensures the minimisation of mis-sorting practices. In Denmark, recycling centres represent the only method for collecting certain waste fractions, for example wood waste and small combustible waste (mixed waste products with no apparent potential for recycling). Recycling centres also contribute largely to the collection of material fractions that are traditionally collected from households: for example, more than one-third of source separated plastic waste is collected at recycling centres (Miljøstyrelsen, 2017; Swaeco Danmark, 2015).

While characterisation studies on municipal waste from households are moderately available, investigations on waste collected at recycling centres are very scarce in the literature. The composition of some individual material fractions has been studied for the UK and Sweden (Krook et al., 2008; Krook and Eklund, 2010b; Resource Futures, 2011; WRAP, 2016), but little documentation exists for other countries, and what is available is mainly in the form of company reports. However, information on the composition of waste from recycling centres is crucial not only for the correct design of recycling processes, but also for the accuracy of environmental modelling results. Indeed, Bisinella et al. (2017c) demonstrated that life cycle assessment modelling results depend strongly on waste characteristics and their role during recycling.

Denmark has committed to the goal of recycling 50% of household paper, cardboard, glass, plastic, wood, metal and food waste by 2022 (The Danish Government, 2013). However, some material fractions exhibit larger challenges than others with respect to recycling. Fractions such as paper, cardboard, glass and metal waste are characterised by a well-developed state-of-the-art European recycling market; for example, paper and cardboard waste are fully integrated in the production of new paper and cardboard products (Zacho et al., 2018). Moreover, European production of glass is expected to include around 80% of post-consumer glass cullet (Hischier, 2007), while steel-making by electric arc furnaces can be fed with 100% steel scrap (Rigamonti et al., 2018). While impurities affect all recyclable fractions to different extents, little knowledge is

Box 1. Recycling centres

Recycling centres are large collection points for municipal solid waste. Private citizens bring their waste individually and sort it into a number of material fractions. Small companies, for example builders, are allowed to bring a limited amount of waste upon the payment of an entrance fee. The numbers of collected material fractions depend on the municipality, but in Denmark it is generally around 30-40: only food waste is strictly prohibited. In addition to traditional materials such as glass, metals and paper, more specific waste fractions exist: for example, plastic waste is subdivided into hard plastic, plastic film and PVC waste; construction and demolition waste is distributed into asbestos, tile, brick, window and door sections. The waste fractions are collected in >10m^3 stationary containers. When the containers are full, they are removed and transported to the company treating the waste. Some containers, e.g. wood waste, are equipped with a compaction unit to increase the waste density and facilitate transport. Almost all waste fractions are routed to recycling. When the waste does not hold recycling potential it is placed either in the “small combustible waste” container (sent to incineration) or in the “waste to landfill” container (landfilled).

available on contamination levels of wood and plastic waste. Furthermore, the potential leaking of recyclable resources into waste streams not destined for recycling is unknown, possibly holding back the achievement of recycling goals. At recycling centres, small combustible waste is the fraction expected to most likely act as a sink for recyclable resources.

1.2 State-of-the-art

Wood waste, plastics waste and small combustible waste were selected as target fractions collected mainly at recycling centres and for which characterisation studies are needed. Furthermore, alternative management solutions may improve recycling and provide larger environmental benefits than are being achieved by current management.

Post-consumer wood waste generation is estimated currently at 15 million tonnes in the EU (Gurria et al., 2017) and is forecast to increase (Mantau, 2015). Recycling of wood waste is promoted in the EU through recycling targets on packaging wood waste and on construction and demolition waste (EC, 2018c). At the same time, European policies encourage the use of wood waste for energy utilisation (e.g. the Renewable Energy Directive), potentially creating tension across sectors, which in turn may result in suboptimal management choices.

Recycling of wood waste currently focuses mainly on particleboard as the main recycling application and is expected to remain so in the near future (FAO/UNECE, 2016; Vis et al., 2016). Particleboard is a type of wood-based panel obtained from pressing wood chips at high temperature into a mat upon the addition of a gluing resin (Wilson, 2010); the mat is then covered with a thin layer of wood, plastic or paper. Due to the simplicity of the process, particleboard can *technically* accommodate almost all wood waste types and qualities, although the large presence of fibreboards may cause processing problems, due to the large amount of resin already included (Daian and Ozarska, 2009). Moreover, impurities may be recycled along with the wood material, thereby potentially posing safety risks (Bergeron, 2014). Alternative recycling applications (floor boards, other wood-based panels, wood composites, pellets, pulp to paper and bio-based chemicals) are restricted by the presence of impurities in wood waste (Winder and Bobar, 2016). Nonetheless, while impurities have been largely documented in wood waste for incineration (e.g. Astrup et al., 2011; Edo et al., 2015, Krook et al., 2004; 2006), their presence in wood waste for recycling is scarcely quantified (Abb et al., 2010; Bouslamiti et al., 2012), despite the potential challenge to a clean recycling industry.

Environmental assessments conducted on wood waste management have often offered contradictory results. While some LCA studies indicate recycling as the best management solution due to maximising resource efficiency (Bais-Moleman et al., 2018) and contributing to climate change mitigation (Börjesson and Gustavsson, 2000; Rivela et al., 2006a), other LCAs indicate energy use as the preferable option as a result of substituting fossil energy sources (Dodoo et al., 2009; Morris, 1996) and preventing the dispersion of pollutants (Werner et al., 2010). With wood waste amounts being increasingly recycled in cascades, there is a general interest in demonstrating that clean and safe recycling loops are reflected in reduced pressure on the environment.

Post-consumer plastic waste generation reached 27 million tonnes in 2016 in the EU, following a steady increase in plastics production, which in the past decades has replaced more traditional materials (Plastics Europe, 2018). The recycling rate of plastic waste in the EU was estimated at 31% for 2016, although values vary considerably country by country (Plastics Europe, 2018). Indeed, the recycling of plastic waste is generally considered more complicated than for other materials (Rigamonti et al., 2015), and the EU has recently released a dedicated strategy to tackle challenges related to its management (EC, 2018a).

Issues in plastic waste recycling arise mainly from the high purity required for reprocessing as opposed to the heterogeneity of the waste. Indeed, the term "plastics" is an umbrella to address a variety of polymeric materials, e.g. polypropylene (PP), polyethylene (PE), polyethylene terephthalate (PET), polyvinyl chloride (PVC) and polystyrene (PS). All plastic types require the use of plasticisers, stabilisers and other additives to contrast the otherwise poor properties; the extent of these additions varies with the application plastics are used into (Jasso-Gastinel and Kenny, 2017; Murphy, 2001; Pitchard, 1998).

The significant heterogeneity of plastic waste causes significant technical barriers (e.g. different polymers cannot be recycled together), requiring advanced pre-treatment operations whose large costs may be reflected in high selling prices that cannot compete with the virgin plastic market (EC, 2018b). Several typologies of sorting/recycling may be employed, depending on the plastic characteristics. However, while limited compositional knowledge is available for plastic waste from households (e.g. Eriksen et al., 2018b; Luijsterburg and Goossens, 2014; Pivnenko et al., 2017) plastic waste from recycling centres are not documented, despite the large contribution of this collection method and despite plastic properties are expected to differ from those collected from households.

While many LCA studies have been conducted on plastic waste management (e.g. Al-Salem et al., 2014, Arena et al., 2003, Benavides et al., 2017, Ferreira et al., 2014, Rigamonti et al., 2014), these studies did not clearly tackle the role of plastics waste quality on the environmental savings from its recycling. Extensive sorting required to remove impurities from plastic waste streams may cause polymer degradation, thereby lowering the quality of the recycled products, which in turn may not fully substitute primary materials. Additionally, lower-quality may not only decrease potential environmental savings, but also make recycling uneconomical. Without detailed information on plastic waste characteristics, a solid evaluation of plastic recycling systems is not possible.

Small combustible waste entails waste products that are not considered recyclable but hold potential for energy recovery. The term “small” refers to products with one dimension smaller than one metre, as opposed to “large” objects, which are typically delivered by small companies. Small combustible waste represents 16% of the total waste collected at recycling centres, making it one of the largest fractions at these sites (Bisinella et al., 2017b).

In principle, small combustible waste should be free from any recyclable materials. However, Krook et al. (2010b) and WRAP (2016) identified it as one of the most contaminated fractions at recycling centres, suggesting the potential leaking of valuable resources to incineration. Indeed, while the contamination of recyclable fractions is a recognised issue, a less discussed problem is when recyclable materials enter waste streams not destined for recycling (Villanueva and Eder, 2014). Non-recovered recyclable fractions cause a loss of recycling potential and, consequently, of environmental savings that would have otherwise been provided by their recycling. However, studies addressing the composition of small combustible waste are not available, with DEFRA (2009) estimating it as the second most significant data gap in waste composition analysis (after littering). Detailed knowledge about the composition of this waste fraction is pivotal in estimating the recycling potential that is lost when small combustible waste is sent to incineration.

1.3 Research objectives

The overall aim of this PhD thesis is to contribute to an improved knowledge on the resource quality of wood waste, plastic waste and small combustible waste collected at recycling centres. The intention is further to investigate the role of the resource quality in the recycling chain of the waste resources, with a specific focus on Danish conditions. This is finalised by providing recommendations toward improved collection and management of the included waste fractions. The research involves the following specific objectives:

- Sampling and characterisation of the selected waste fractions from Danish recycling centres with respect to material composition, physico-chemical characteristics and contamination level, in order to provide insights into the resource quality of these fractions with respect to recycling (Faraca et al., **I, III, V**)
- Evaluating the recyclability of the selected waste fractions based on their characteristics in terms of recycling indicators, recycling efficiencies and cascading potentials for the identification of crucial aspects of the overall recycling chain (Faraca et al., **II, III, IV, V**)
- Establishing and implementing a consistent modelling framework for quantifying environmental and financial consequences provided by improved recycling through the integration of material flow analysis, life cycle assessment, life cycle costing and global sensitivity assessment, in order to develop a solid scientific basis for decision making (Faraca et al., **II**, **IV**, **V**)

The synopsis part of this PhD thesis is structured as follows:

Section 2 provides an overview of the methodological approaches adopted in the sampling, characterisation and modelling of the selected waste fractions.

Section 3 presents the results on the composition of wood waste with respect to the composition of material and chemical impurities. These results are used to formulate alternative management solutions for wood waste, which are then evaluated for environmental impacts.

Section 4 discusses the results of the plastics waste composition, focusing on the role of impurities in the recycling chain. Alternative recycling routes differing in the approach of handling impurities are evaluated in terms of environmental and financial impacts.

Section 5 describes the material composition of small combustible waste and estimates the loss of potential to contribute toward increased recycling rates as well as increased environmental savings.

Section 6 summarises the key conclusions of the thesis and suggests recommendations for enhanced recycling systems for wood waste, plastic waste and small combustible waste.

Finally, **Section 7** offers future perspectives.

2 Approaches

2.1 Sampling of waste material fractions

Sampling is the core part of any composition and laboratory analysis. Waste is heterogeneous by definition; taking waste samples that maintain unchanged the properties of the whole lot ("representative sampling") involves a specific mass- and size-reduction methodology. This ensures that the results of the analyses run on subsamples can be considered valid also for the population from which the subset was taken. Given the large volumes of waste involved at recycling centres, the sampling analyses carried out in Faraca et al. (**I**, **III**, **V**) started by unloading one container (i.e. the sample) onto the ground and performing representative mass reduction (subsampling), as summarised in Table 1. The subsampling techniques included one-dimensional (1D) splitting or bag splitting, according to the specific characteristics of the waste fractions (Edjabou et al., 2015; Gy. 1998; Minkkinen, 2004). In the case of chemical analysis (Faraca et al., **I**), additional mass- and size-reduction steps were performed in order to obtain a homogeneous fine powder.

2.2 Characterisation of waste material fractions

The characterisation of wood waste, plastic waste and small combustible waste involved multiple classification bases covering several characteristics of the waste products, such as application, quality, colour, expected lifetime and chemical composition (Table 2). The sample level (primary sample, subsample or lab sample) at which the investigation is carried out depended on the property under assessment: while product application was assessed at the primary or the subsample level, chemical analyses focused on the lab samples.

Waste fractions were characterised with respect to the presence of impurities, in order to capture their potential consequences on the quality of recycled materials. Although impurities can be classified generically into physical (material) and chemical impurities, specific characteristics depend on the waste fraction under assessment (see Box 2). The presence of material impurities was quantified by manual separation for wood and plastic waste (Faraca et al., **I**, **III**); however, in the case of small combustible waste, impurities were interpreted as the recyclable materials entering waste streams not destined for recycling (lost recycling potential; Faraca et al., **V**). Chemical impurities were analysed in wood waste (Faraca et al., **I**) and compared to levels in pre-consumer wood (sampled from Danish retailers and processed with the same methodology adopted for wood waste).

Box 2. Material vs chemical impurities

Material impurities – in principle - are mechanically separable and can be classified, according to Faraca et al. (**I**), into:

- Misplacements, i.e. foreign material fractions and untargeted products not matching the sorting guidelines. Apart from easily recognisable foreign products (e.g. metal cans in wood waste or plastic waste), misplacements include: impregnated wood in wood waste; plastic films in hard plastic waste and vice versa; recyclable fractions in small combustible waste.
- Interfering materials, i.e. untargeted material fractions attached to the targeted waste material (part of the product). These include metal nails in wooden desks or upholstering in wooden furniture for the case of wood waste; metal parts in plastic toys or paper labels in plastic bottles, as well as multi-polymer products (two or more homopolymers assembled together, i.e. a HDPE cap in a PP bottle) for the case of plastic waste.
- Lower-quality materials, i.e. targeted material fractions with lower properties. These materials can *technically* be recycled (up to certain thresholds) but would lower process yields. Actual contaminating products depend on the targeted waste fraction and on the final application of the recycled product. Examples include: wood waste of quality grade Q3 (see Table 2) for the case of wood waste; coloured plastics and Low Quality applications (see Table 2) for the case of plastic waste.

Chemical impurities include organic and inorganic compounds on the surface of or bound to the material fraction targeted for recycling. Examples can be the presence of pentachlorophenol in wood waste (e.g. due to anti-decay treatment) or the presence of colorants, heat stabilisers and flame retardants in plastic waste. Chemical impurities in recycled products are often unavoidable, and may pose safety risks when exceeding certain thresholds (Bilitewski, 2012).

Table 1. Overview of the methodology for sampling and processing of samples adopted for wood waste, plastic waste and small combustible waste. More details in Faraca et al. **I**, **III**, and **V**.

Phase	Wood waste (Faraca et al., I)	Plastic waste (Faraca and Astrup, III)	Small combustible waste (Faraca et al., V)
Waste material fraction(s) sampled[1]	One fraction: Clean wood waste for recycling	Three fractions: Hard plastic waste, Plastic film waste and PVC waste	One fraction: Small combustible waste
Number and municipality of sampling sites	Three: Middelfart, Gelsted and Glamsbjerg	Three: Copenhagen, Roskilde, Silkeborg	Eight: Tandskov, Tietensgade, Lemvig, Viby, Jyllinge, Hvidøvre, Høje Taastrup
Primary samples. Waste quantity object of the investigation, for each sampling site	One container (~2-3 tonnes)	One container per material fraction (hard plastic ~1 tonne; plastic film ~200 kg; PVC ~500 kg)	One container (~2-4 tonnes)
I Subsampling (to "subsamples"). Mass reduction of primary samples to subsamples	Not applicable	Hard plastic waste: split into two halves by an armed truck; one half represents the subsample. Plastic film waste: 1D splitting[2] until reaching of ~100 kg PVC: not applicable	Not applicable
I Size reduction	Shredding on site	Not applicable	Not applicable
II Subsampling (to "lab samples"). Mass reduction of primary or sub- samples to lab samples	1D splitting[2] until reaching of 10-15kg	Hard plastic waste: bag splitting[3] until reaching of 40-50 kg Plastic film waste: bag splitting[3] until reaching of 40-50 kg PVC: not applicable	Bag splitting[3] until reaching of 40-50 kg
Further processing. Size- or mass-reduction of lab samples	Grinding by use of disc mill, splitting by riffle splitter until reaching ~ 300g of powder	Not applicable	Not applicable

1 Reflects the naming standards of the municipality where the sampling was carried out

[2] Taking increments (alternatively saved or discarded) from an elongated flat pile with two parallel cut-off surfaces. Saved increments represent a subsample

[3] Filling of 120L paper bags which were alternatively saved or discarded

Table 2. Overview of the classification bases and related methodology for characterisation of samples of wood waste, plastic waste and small combustible waste. See details in Faraca et al. (**I**, **III**, and **V**). ATR: attenuated total reflectance; FTIR: Fourier-transformed infrared; GC-MS: gas chromatography mass spectrometry; ICP-OES: inductively coupled plasma optical emission spectrometry; ICP-MS: inductively coupled plasma mass spectrometry; MAE: microwave assisted extraction; PAH: polycyclic aromatic hydrocarbons; PCB: polychlorinated biphenyls.

Classification basis	Wood waste (Faraca et al., I)	Plastic waste (Faraca and Astrup, III)	Small combustible waste (Faraca et al., V)
Material fraction	One material fraction: Wood waste	One material fraction: Plastic waste	Twelve material fractions: 1) Glass, 2) Paper, 3) Cardboard, 4) Metals, 5) Plastics, 6) Wood, 7) Electrical and electronics, 8) Textiles, 9) Garden waste, 10) Combustibles, 11) Non-combustibles 12) Hazardous waste.
Product application. Application of the waste products in the samples	Five categories: • Off-cuts • Packaging • Construction and demolition • Furniture • Misplacements. Assessed by visual assessment at the primary sample level	Eight categories: • Food packaging (FP) • Non-food packaging (NFP) • Electrical and electronics (EE) • Pharmaceuticals (PH) • Toys (T) • Construction (C) • Automotive (A) • Other (O) Assessed by visual assessment at the primary sample (PVC) or subsample (hard plastic and plastic film) level	Different application list depending on the material fraction (see Table 1 in Faraca et al., **V**). Assessed by visual assessment at the primary sample (material fractions 1, 4, 5, 8, 10) or subsample (material fractions 2, 6, 7, 12) level.
Type of products. Detailed description of waste products, e.g. box, bottle, etc.	Fifteen classes (see Table 1 in Faraca et al., **I**). Assessed by visual assessment at primary sample level	Number of classes depends on application (see Appendix F in Faraca and Astrup, **III**). Assessed by visual assessment at primary sample (PVC) or lab sample (hard plastic and film) level	Not applicable
Quality. Property of the waste products to be	Four classes[1]: • Q1, untreated wood waste	Three classes[3]: • High Quality, incl. FP applications	Not applicable

Classification basis	Wood waste	Plastic waste	Small combustible waste
recycled to a certain application use	• Q2, relatively treated wood waste used in indoor applications • Q3, wood waste treated with non-hazardous compounds, i.e. outdoor applications[2] • Q4, hazardous wood waste Assessed by visual assessment at primary sample level	• Medium Quality, incl. EE, PH and T applications • Low Quality, incl. NFP, C, A and O applications Assessed by visual assessment at primary sample (PVC) or subsample (hard plastic and plastic film) level	
Physical properties	Not applicable	<u>Colour</u>: three classes (transparent, black-coloured and other colours plastic waste) <u>Expected lifetime</u>: three classes (shorter than 1 year, between 1 and 10 years, longer than 10 years) Assessed by visual assessment at the primary sample (PVC waste) or subsample (hard plastic and plastic film waste) level	Not applicable
Chemical properties	Concentration of 65 inorganic elements assessed at lab sample level by ICP-MS or ICP-OES, depending on the element. Concentration of 27 PAHs, 7 PCBs and 15 phenols and assessed by MAE followed by GC-MS. The analysis was performed also to 7 samples of pre-consumer wood, sourced from Danish retailers and processed as described in Table 1	Seven polymer classes: PP, PET, HDPE, LDPE, PS, PVC, engineered polymers. Assessed by FTIR analysis (4300 Agilent Technology equipped with an ATR diamond lens) at the lab sample level (only hard plastic and plastic film waste)	<u>Material fraction 4</u> (metals): two classes (ferrous, non-ferrous). Assessed by magnetic separation at the lab sample level. <u>Material fraction 5</u> (plastic): seven polymer classes (PP, PET, HDPE, LDPE, PS, PVC, engineered polymers). Assessed by FTIR analysis equipped with an ATR diamond lens at the lab sample level.

[1] there is no internationally agreed classification for wood waste, but European recyclers typically refer to the German legislation (Altholz V, 2012).

[2] Additionally, in UK facilities a limit is set on acceptance of fibreboard at 5-10%. In Faraca et al. (**I**), fibreboards were addressed as Q3 grade

[3] according to Eriksen et al., 2018

2.3 Recyclability of waste material fractions

The recyclability of waste material fractions defines the resource efficiency and depends on the method of collection, the waste material composition and the type of sorting/reprocessing plant. Recyclability is generally quantified in terms of recycling indicators (or rates) (Espinoza and Soulier, 2018). Recycling rates express the quantity of recyclable waste sent to sorting (CR-R; i.e. collected for recycling), sent to recycling (RR; i.e. recycling rate) or recycled (EoL-RR; i.e. end-of-life recycling rate) with the total quantity of recyclable waste generated (Figure 1). These indicators are also used at the EU level as a metric for monitoring countries performances (Box 3). While CR-R is typically obtained by collecting data from e.g. national statistics, RR and EoL-RR can be calculated from CR-R by considering sorting and recycling efficiencies (or losses).

For each specific recyclable material and treatment technology, sorting and reprocessing efficiencies can be measured at the facilities, derived from available publications and reports or calculated based on the characteristics of the waste input flow. In Faraca et al. (**V**), independent scientific publications were reviewed to estimate typical (i.e. average) sorting and reprocessing efficiencies for each recyclable waste material under assessment. In Faraca and Astrup (**III**) a material flow analysis was carried out, assigning individual sorting and reprocessing mass transfer coefficients (TCs) to the plastic waste products grouped by properties relevant for sorting/reprocessing (in this case the presence of coloured products and impurities; Table 3), upon knowledge of their proportions in the waste. The proportions were calculated from the characterisation results for each polymer and the quality class of plastic waste.

Box 3. Recycling targets

In the EU, countries efforts to improve the resource efficiency of materials are quantified and monitored by setting binding minimum recycling rates (recycling targets). Such targets (e.g. on municipal waste, packaging waste and waste electrical and electronic equipment and) act as the main drivers towards a circular economy, as countries not complying with existing legislation are subjected to fees. Recycling targets are increased cyclically through European directives. However, recycling targets have been criticised for being quantity-based and for the unclear directions on the calculation method, which may represent waste flows “collected for recycling” (but not necessarily recycled). The latest amendments (EU, 2018a; 2018b) updated the calculation method, reflecting in recycling targets waste flows entering a recycling process (after sorting of impurities).

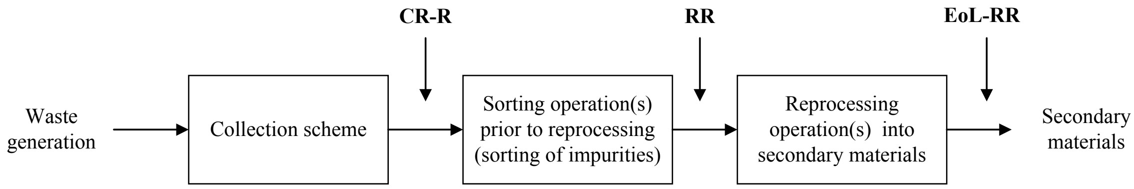

Figure 1. Recycling indicators definition with respect to recycling treatment phases. CR-R: collected for recycling rate; EoL-RR: end-of-life recycling rate; RR: recycling rate. Adapted from Faraca et al. (**V**).

2.4 Life cycle assessment modelling

Life cycle assessment (LCA) is a standardised methodology employed for the comparative assessment of environmental impacts of systems (ISO, 2006a; 2006b). LCA is used widely in waste management to support decision making because it involves a holistic approach that takes into consideration direct (i.e. emissions directly related to the system assessed) and indirect impacts (i.e. effects induced on other sectors), thus avoiding problem-shifting (Wenzel et al., 1997). It consists of four phases: 1) goal and scope definition, 2) inventory analysis, 3) impact assessment and 4) interpretation. Three LCA studies are reported as part of this thesis, addressing the management of wood waste, plastic waste and small combustible waste, respectively (Faraca et al., **II, IV** and **V**).

Table 3. Transfer coefficients (TCs) assumed for plastic waste products, grouped by presence of coloured products, misplacements and interfering materials. From Faraca et al. (**III**).

Waste product	TCs	Comment
(Optical) sorting to homopolymer reprocessing line		
Black-coloured plastic	0	Black plastic cannot be recognised by infra-red technology
Non-plastic interfering materials	1	Assumed sorted together with the main polymer
Copolymers	0.5	Partially sorted to one of the polymers because having a similar spectrum
Multi-polymer products	0.5	Partially sorted to the homopolymer constituting the majority of the product
Misplacements	0	Not detected as plastic
Other plastic products	1	Sorted to homopolymer line
Reprocessing to recycled materials		
Non-plastic interfering materials	0	Rejected by separation equipment
Copolymers	0	Rejected during floating/extrusion
Multi-polymer products	0	Rejected during floating/extrusion
Other plastic products, clear	1	Recycled
Other plastic products, coloured	<1	Actual value depends on the polymer and comes from available literature

2.4.1 Goal and scope definition

In this first phase of LCAs, the study is defined with respect to the intended aim and the decision-context. The LCA studies presented in this thesis employed the knowledge obtained from the characterisation of wood waste, plastic waste and SCM waste (Faraca et al., **I, III, V**) to develop management alternatives for the waste fraction under assessment. The comparison of such management alternatives against the baseline case represented the goal of the individual LCAs (see Table 4). All case studies presented in this thesis followed a consequential modelling approach, which seeks to identify the consequences of a decision (i.e. an additional unit of a product or service placed in the market; see Box 4). Conformingly with a consequential approach, multi-functionality of processes was handled with substitution by system expansion (EC-JRC, 2010). This requires identifying the affected technologies/products.

The central element of an LCA is the functional unit (FU), which is the metric describing the service provided by the system under assessment (ISO, 2006a). In LCAs of waste systems, the service is often expressed per unit input (e.g. one tonne; Cherubini et al., 2009). The composition of the FUs in Faraca et al., (**II, IV, V**) reflected the characterisation findings from Faraca et al., (**I, III, V**).

Box 4. Attributional vs consequential modelling

Two LCI modelling approaches can be implemented in the context of LCA: attributional and consequential modelling. While attributional modelling depicts the environmental impacts of a system over its life cycle stages, consequential modelling strives to capture the environmental consequences of a decision (EC-JRC, 2010). Attributional modelling describes the system as it is, was, or is forecast to be (Tillman, 2000). Conversely, consequential modelling does not represent the actual or forecast system, but hypothesises the system reacting to a change (e.g. taking into consideration market mechanisms, political interactions and consumer behaviour; Weidema, 2003). While in the attributional case the system is isolated into a static technosphere, in consequential modelling the system interacts with dynamic markets/other systems (Weidema et al., 2009). The two modelling approaches solve the case of multifunctional processes (i.e. processes providing more than one service such as recycling and incineration) in different ways: while attributional modelling typically allocates emissions to each function on a certain basis (e.g. mass or price), consequential modelling uses system expansion to make the system comparable, i.e. by adding a not provided function or by subtracting a not required function and substituting it by the one replaced. (EC-JRC, 2010).

Table 4. Overview of the relevant methodological assumptions for the consequential LCAs conducted in the papers forming the basis of this thesis (Faraca et al., **II**, **IV**, and **V**)

Assumption	Wood waste (Faraca et al., II)	Plastic waste (Faraca et al., IV)	Small combustible waste (Faraca et al., V)
Goal	To compare the environmental impacts from current practices (mixed qualities) of post-consumer wood waste recycling cascades with alternative cascading possibilities (separation of qualities)	To compare the environmental impacts from alternative recycling routes for post-consumer hard plastic waste	To compare the environmental impacts from recycling the "recyclable materials" correctly sorted from SCM with the alternative case of incineration.
LCA approach	Consequential	Consequential	Consequential
Functional unit (FU)	The management of 1 tonne of post-consumer wood waste collected at Danish recycling centres with the quality composition according to Faraca et al., (**I**):4% Q1, 56% Q2, 32% Q3, 6% Q4, 2% other material fractions	The management of 1 tonne of post-consumer hard plastic waste collected at Danish recycling centres with the composition according to Faraca et al., (**III**): 53% PP, 19% PE, 6% PVC, 2% PET, 1% PS, 10% engineered polymers, 9% other material fractions	The recycling of the yearly amount of recyclable waste that could be recovered by correctly separating SCM at Danish recycling centres (i.e. 144,845 tonnes of recyclable waste with the composition according to Faraca et al., **V**)
Geographical scope	Denmark	Denmark	Denmark
Temporal scope	2015-2030	2017-2030	2018-2050
System boundaries	Start after the collection phase and end with final management (incineration/landfill) of the FU	Start after the collection phase and end at the point of substitution of secondary materials/energy or final management of any residues	Start after the collection phase and end at the point of substitution of secondary materials/energy or final management of any residues
Scenarios	Fifteen scenarios, divided into two sets: "Wood waste, mixed qualities": four scenarios (A1-A4) where all wood waste qualities perform up to 4 recycling to particleboard cascades "Wood waste, quality distinction":eleven scenarios where Q1+Q2 are recycled and Q3+Q4 are incinerated. Recycling applications are:	Three scenarios: "simple mechanical recycling (sMR)": simple sorting of PP and PE and reprocessing to secondary pellets "advanced mechanical recycling (aMR)" multi-step sorting of PP, PE, PET and PS and reprocessing to secondary pellets	Two scenarios: "recycling": each recyclable fraction is recycled according to average European processes "incineration": municipal solid waste incinerator with energy recovery

Assumption	Wood waste	Plastic waste	Small combustible waste
	particleboard (B1-B4) floorboard (C1-C4) wood insulation board (D1) wood plastic composite (E1) wood pellets (F1)	"feedstock recycling (FR)": pyrolysis of PVC-freed FU to produce secondary oil and steam	
Marginal energy	Electricity: 61% wind energy, 39% biomass-energy Heat: natural gas	Electricity: 61% wind energy, 39% biomass-energy Heat: natural gas	Electricity: 56% wind energy, 28% biomass-energy, 16% natural gas Heat: natural gas
Marginal products	Scenarios A1-C4 and E1: corresponding products from virgin wood Scenario D1: glass wool insulation Scenario F1: energy In all scenarios metal residues are recycled, ashes and inert residues are landfilled, other residues are incinerated	Scenarios sMR and aMR: corresponding homopolymer pellets from virgin plastic Scenario FR: pyrolysis oil substitutes crude oil and pyrolysis steam substitutes natural gas In all scenarios metal residues are recycled, ashes and inert residues are landfilled, other residues are incinerated	Material fractions 1-7: corresponding products from virgin sources Material fraction 8: polyurethane flexible foam Material fraction 9: mineral fertilisers and peat In all scenarios metal residues are recycled, ashes and inert residues are landfilled, other residues are incinerated
Assessment method	IPCC (IPCC, 2013)	ILCD recommended (EC-JRC, 2011)	ILCD recommended (EC-JRC, 2011)
Impact categories	Global Warming Potential 100 years; Global Warming Potential 500 years	Global warming potential 100 years, Ozone depletion; Human toxicity (cancer effects); Human toxicity (non-cancer effects); Particulate matter; Ionising potential 100 years; Photochemical oxidation formation potential; Terrestrial acidification; Terrestrial eutrophication; Eutrophication freshwater; Eutrophication marine; Ecotoxicity; Depletion of resources	Global warming potential 100 years, Ozone depletion; Human toxicity (cancer effects); Human toxicity (non-cancer effects); Particulate matter; Ionising potential 100 years; Photochemical oxidation formation potential; Terrestrial acidification; Terrestrial eutrophication; Eutrophication freshwater; Eutrophication marine; Ecotoxicity; Depletion of resources

Geographical and temporal scopes specify the *where* and *when* the system is evaluated, i.e. the conditions for which the LCA results can be considered valid. Their choice should be stated transparently in order to avoid the application of results to contexts beyond the scope of the LCA. Temporal and geographical scopes influence the selection of the scenarios, the affected technologies (referred to as the *marginals* in consequential LCA), energy and products, and the data quality requirements (EC-JRC, 2010). Mid- to long-term scopes were considered in Faraca et al. (**II, IV, V**).

System boundaries define the life cycle stages included in the systems under assessment. As common practice in waste LCAs, the studies presented in this thesis adopted the "zero burden" assumption, i.e. the waste fraction is considered free from any impacts caused or saved during its former use (Weidema et al., 2009).

The identification of the marginal technologies, energy and products is specific to the consequential methodology, as these *marginals* represent the unit processes capable of reacting as a consequence of a change in the demand or supply of a specific product/service (the term "marginal" indicates that a small change is considered; Weidema et al., 2009). Their choice depends on the geographical and temporal scope and is of crucial importance, as it is likely to influence the magnitude of the impacts/savings of the systems. These marginals are identified by analysing market trends, energy systems and policy targets, considering possible bans or constraints (EC-JRC, 2010). In Faraca et al. (**II, IV**, **V**), the marginal electricity was identified as the energy technology expected to increase in capacity within the temporal scope assumed in the studies, according to Muñoz et al. (2015): due to the commitment to ambitious renewable energy targets, marginal electricity for Denmark was assumed almost fossil-free. Conversely, the choice of marginal heat is constrained by the capacities and infrastructure related to district heating (Fruergaard et al., 2010), and in Denmark it is generally assumed produced from natural gas boilers, although in the long term it is expected to be replaced gradually by renewable energies (Schmidt et al., 2016). Finally, marginal products substituting for the recycling of materials were identified mainly with the corresponding products from primary materials. Although it can be argued that for some material fractions primary production already includes secondary resources (given the large availability in recycled materials), this cannot be assumed in a consequential LCA, as residues cannot adapt their market to respond to a system change (Weidema et al., 2009), being "constrained" by definition. Due to the high related uncertainty, the marginals were tested by modelling alternative assumptions (see Section 2.4.4).

The goal and scope phase must also clarify the selection of impact categories and the assessment method employed to translate environmental exchanges into impacts. Impact categories represent environmental issues of concern which are quantified by the LCA for the system under assessment (ISO, 2006b). In Faraca et al. (**IV**, **V**) results were provided for 13 impact categories (see Table 3), but interpretation focused mainly on global warming potential (GWP), because this impact category is based on widely accepted concepts. Climate change mitigation has also become a political priority (to which to commit through ambitious targets on renewable energy), due to very large human emissions of greenhouse gases and the social pressure from recently observed climate change evidence (Steffen et al., 2015). The environmental assessment of wood waste (Faraca et al., **II**) tackled critical methodological issues in LCAs of biomass systems (Box 5). Conforming to the latest research findings, the modelling approach accounted for all emissions at the moment of release (dynamic LCA), including carbon dioxide from biogenic sources. Moreover, impacts from indirect land use change (iLUC) were addressed.

2.4.2 Life cycle inventory (LCI)

In this phase all input and output data related to unit processes are collected, and all related environmental exchanges are listed. Data can be primary (from measurements and experiments) or secondary (from independent studies and databases); it is paramount that, when used, secondary data are consistent with the scope of the study (EC-JRC, 2010). Data can be single values or a distribution of values. The latter case allows assigning an uncertainty to the parameter, which can be evaluated in the interpretation phase of the LCA. The modelling presented in this thesis relied mainly on secondary data. Their collection involved careful review of a large number of studies and databases: a probability distribution could then be assigned to each model parameter, defining the baseline value used in the modelling (the median of the dataset) and the associated uncertainty.

2.4.3 Life cycle impact assessment (LCIA)

In this phase, all emissions inventoried during the LCI for the system under assessment are converted into environmental impacts by using characterisation factors specific to the impact category and the substance emitted. In Faraca et al. (**II, IV**, **V**) the LCA software EASETECH was used for the computation

Box 5. Critical issues in Global Warming Potential assessment.

GWP measures the potential effects of a GHG emission on the climate compared to CO_2 over a certain time horizon (TH) (IPCC, 2013). Three fundamental aspects related to GWP accounting are generally overlooked in LCA studies. First, emissions of biogenic CO_2 from biomass combustion are not accounted for, because they are assumed to be re-absorbed as new biomass grows (i.e. climate neutral). However, the behaviour (decay) of CO_2 in the atmosphere is the same irrespective of its origin, and recent studies (e.g. Cherubini et al., 2011) have demonstrated that biogenic CO_2 emissions should be considered when originating from the combustion of biomass species with a rotation period of several decades (e.g. wood forests or wood waste). The behaviour of biogenic CO_2 emissions can be modelled by combining its atmospheric decay curve with the CO_2 sequestration by plantation growth, according to Cherubini et al. (2011). The extent of the sequestration depends on the biomass species, i.e. its rotation period.

The second limitation of GWP is the inconsistency between the definition of TH and its accounting method. While THs are defined as a fixed time for integration of the effects of the CO_2 (typically 100 years from the year 0, but in principle any time period), emissions occurring during the life cycle are generally modelled traslated to the year 0 and then aggregated (irrespective of the actual time at which they occur), thus making the TH move in time (100 years from the moment the emission occurs). Considering a system starting in 2015 (year 0), if an emission occurs in 2035, but is translated to the year 0 (i.e. modelled as occurring in 2015), impacts for TH=100 years are in reality calculated for 2135, which falls outside the defined TH. This can be solved by assigning to each emission the year on which it occurred and by calculating the integral over a fixed TH, according to Levasseur et al. (2010).

Finally, GWP assessments, from a consequential modelling perspective, do not typically include impacts from indirect land use change (iLUC). iLUC accounts for the upstream consequences (transformation of land somewhere else, even beyond the assessed geographic system) of demanding the land for growing biomass (wood or crop) plantations due to a change in demand for the biomass products. According to Schmidt et al. (2015), the iLUC impacts to grow a forest are a function of the amount of land demanded and the type of land transformation. Avoiding (e.g. by recycling and displacing virgin material) an additional demand for wood products should thereby be credited with the corresponding iLUC CO_2 impact avoidance.

(Clavreul et al., 2014). The characterised results can possibly be normalised (i.e. made comparable across impact categories by expressing them in a common unit) and weighted (i.e. aggregated across impact categories based on value-choices); these were not applied in any of the LCAs presented herein.

2.4.4 Life cycle interpretation

This LCA phase presents the results of the assessment and identifies the main processes, parameters and assumptions governing the results. The model could possibly be improved iteratively in light of the knowledge attained in this phase. A contribution analysis is used widely to identify significant processes or hotspots. Then, the influence of parameters and assumptions on the variability of the results must be assessed. In Faraca et al. (**II**, **IV**, **V**), the robustness of the model and results to the parameters was tested by global sensitivity analysis (Bisinella et al., 2017). This methodology combines sensitivity analysis (evaluating how sensitive results are with respect to individual parameters) and uncertainty propagation (quantifying the uncertainty in the results due to input uncertainty), enabling the identification of the main parameters contributing to the variability of the results. On the other hand, the robustness of the model and results to system assumptions (e.g. type of technology, energy provision, products) was addressed by scenario analysis, i.e. calculating the results for alternative key assumptions.

2.5 Life cycle costing

While LCA is accepted as scientific support for decision making, most real-life decisions are constrained by economic considerations. Therefore, when assessing the consequences of a change in the system, a financial evaluation is also necessary. Environmental life cycle costing (ELCC) methodology can be implemented to endow LCA with the financial dimension, provided that the same modelling framework is adopted (goal and scope must be consistent) (Hunkeler et al., 2008). In the financial part of an ELCC, all costs incurred by all stakeholders acting in the system under assessment are included, albeit monetary flows are considered from the point of view of one stakeholder (e.g. the collector, the recycler, citizens, etc.) (Swarr et al., 2011).

In Faraca et al. (**IV**), an ELCC was carried out, expanding the LCA on alternative recycling routes for plastic waste (as presented in Section 2.4) through the evaluation of all monetary flows associated with the FU, assuming that all services included in the system were provided by a single actor. The financial assessment included budget costs (capital costs, such as expenditure for capital goods and their maintenance, and operational costs, such as workers' salaries and the consumption of energy and materials) and transfers (taxes, subsidies and fees applied on the life cycle stages excluding VAT, which is typically recovered by non-household actors, and corporate income tax, since waste operators in Denmark are typically publicly owned), according to Hunkeler et al. (2008). The methodology described in Section 2.4 (collection of data, software employed, and interpretation phase) was applied also to the financial part.

Box 6. Types of life cycle costing (LCC)

In addition to Environmental LCC, Conventional LCC and Societal LCC can also be performed. Conventional LCC represents traditional financial assessments carried out by private companies, while Societal LCC monetises environmental and social impacts. Conventional LCC is typically used as a stand-alone indicator (without an LCA), while Societal LCC internalises LCA and assigns prices to individual emissions, albeit future damages remain unassessed for many environmental emissions due to lack of information (Martinez-Sanchez et al., 2015).

3 Wood waste

This section describes the results of the characterisation of the physico-chemical properties of wood waste for recycling and underlines their relevance for improving the quality of the wood waste and for evaluating the environmental impacts of its management.

3.1 Wood waste material composition

The product application composition of wood waste collected at recycling centres comprised mainly *Construction and demolition* (C&D, a median of 52% of total wood waste, with a standard deviation of 13%) and *Furniture* (25% ±2%) applications, with a smaller contribution made by *Packaging* (17% ±13%) and a marginal share of *Off-cuts* (3% ±5%; Figure 2a).

The assessment of material impurities revealed that wood waste that could be recycled according to the strict definition (i.e. within grades Q1 and Q2, as defined by German legislation, cfr. Table 2) was between 41% and 87%, depending on the recycling centre. The amounts and types of impurities varied in line with application category: *C&D* waste was the most contaminated category, with *Lower-quality materials* making 65% of the category (Figure 2b). *Off-cuts* and *Furniture* were relatively affected by presence of impurities, whereas *Packaging* was the cleanest category. This means that wood waste should not be regarded as a single entity; conversely, by knowing the application composition of the wood waste it would be possible to deduce its level of contamination, prioritising clean material streams for recycling. Overall, *Lower quality materials* was the largest type of impurity, contributing to 78% of material impurities and 29% of total wood waste. Since recycling into particleboard (the main recycling application of wood waste) is a simple mechanical process (wood waste is simply chipped and glued together), this contamination is likely not to be removed by mechanical sorting equipment, corrispondingly considering that wood waste is crushed at recycling centres to increase container density, therefore facilitating transport but hampering the identification of wood qualities. Unsorted *Lower-quality materials* may affect recycled products by causing processing issues (thereby increasing material losses) and potentially incrementing the content of chemicals. The presence of impregnated wood (herein representing 33% of *Misplacements* and 6% of wood waste) may have similar consequences on concentrations of chemicals.

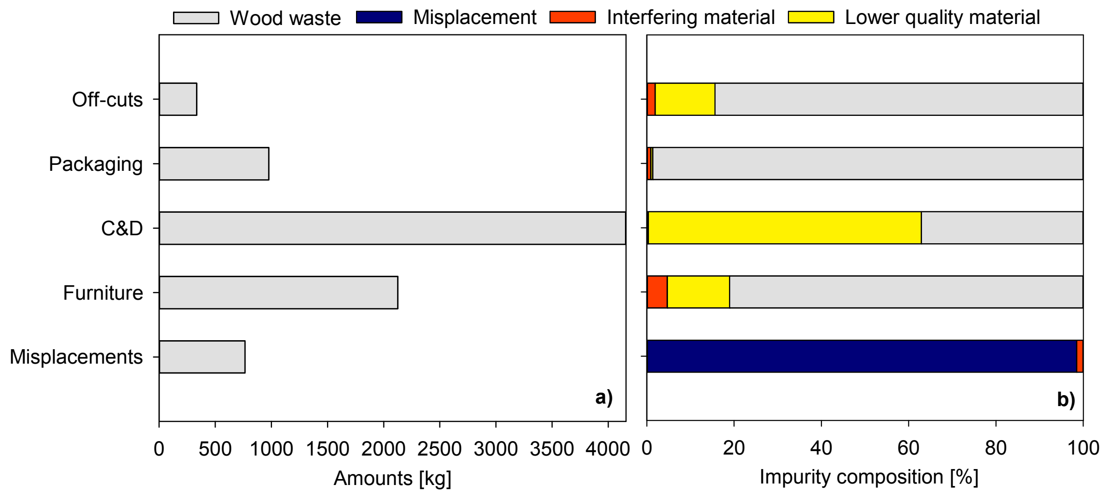

Figure 2. a) Composition of wood waste collected at recycling centres (total quantity of the three sampling sites, expressed in kg wet weight) in terms of product application (see Table 2). b) Composition of material impurities in wood waste collected at recycling centres expressed in % wet weight (see Box 2). Adapted from Faraca et al. (**I**).

3.2 Chemical impurities in wood waste

The concentration of some chemical elements (e.g. As, Pb, PCB, see Figure 3) was observed to vary in line with the wood application category and was always higher than concentrations in pre-consumer wood, probably owing to the fact that these chemicals were added to the wood product to serve a certain scope which is specific to certain applications (e.g. As used to be a widely used preservative in impregnated wood). Most contaminated categories were *Misplacements*, *Furniture* and *Off-cuts*; *C&D* samples showed a variable trend, attributable to the presence of both clean indoor construction materials and polluted products from demolition activities; *Packaging* was overall the cleanest application category. On the other hand, the concentration of some other elements (e.g. Cd, Mg, Mn) was overall similar across the wood waste categories including pre-consumer wood samples, as a result of their extensive use in pigments, paints, and fastening systems (e.g. Cd in inorganic pigments), which are typically found in all wood categories. The chemical levels in wood waste exceeded the allowed concentration of chemicals in recycled particleboard (EPF, 2014; blue horizontal lines in Figure 3) in selected samples with respect to Pb, PCP and PAH, and in the majority of samples as regards Cr and Cu (the average concentration was above the limit); thresholds for As and Cd were never exceeded. This suggests that recycling the entire amount of collected wood waste may lead to safety risks, requiring dilution with clean primary materials, which in turn would decrease the substitution potential of the wood waste.

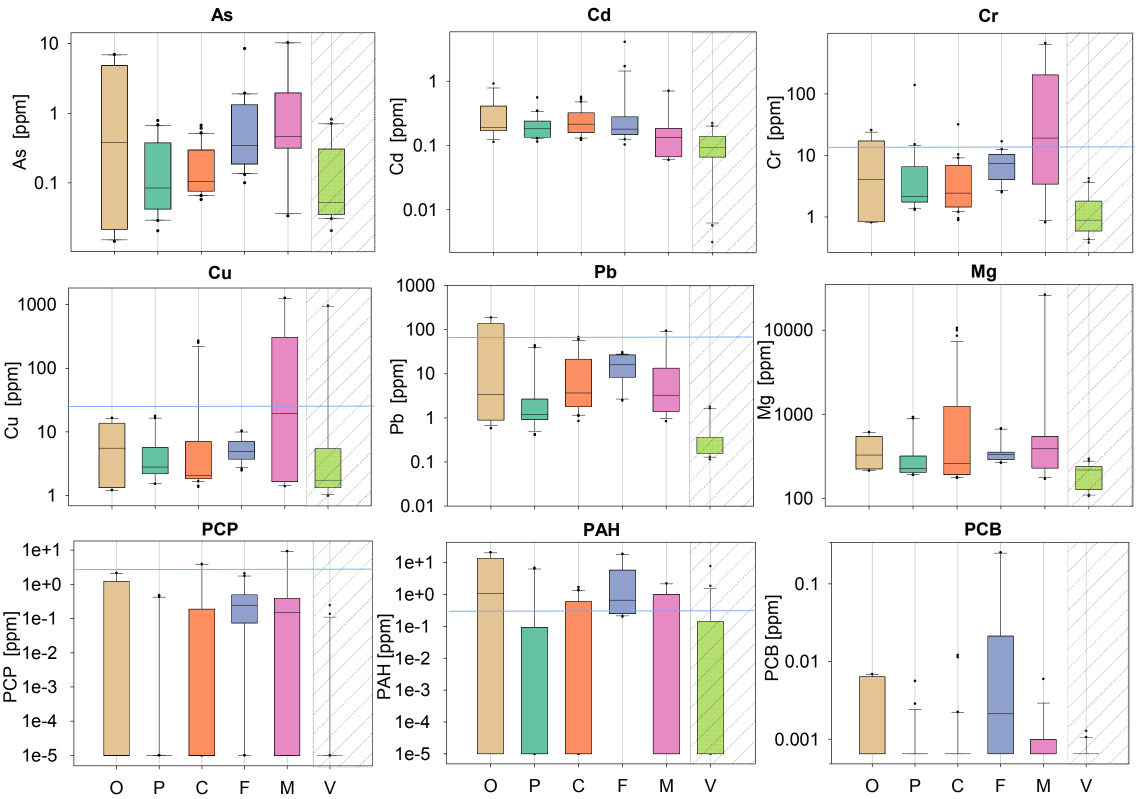

Figure 3. Concentration (part per million, dry weight) of As, Cd, Cr, Cu, Pb, Mg, PCP (penta-chlorophenol), PAH (polycyclic aromatic hydrocarbons, total sum) and PCB (poly-chlorinated byphenils, total sum) in samples of wood waste. The pattern area distinguishes results for pre-consumer wood samples. O: *Off-cuts*; P: *Packaging*; C: *C&D*; F: *Furniture*; M: *Misplacements*; V: *Pre-consumer wood*. Blue horizontal lines indicate the threshold for chemical concentrations in recycled particleboard (EPF, 2014). Adapted from Faraca et al., **I**

Grouping the concentrations of chemicals according to the wood waste quality classes (Q1+Q2: suitable for recycling; Q3: suitable for incineration; and Q4: hazardous wood waste, see Table 2), an increasing trend could be observed from the upper to the lower grades (Figure 4). Concentrations in Q3 and Q4 could be from three-fold (e.g. for As, Pb) up to 26-fold (for Cu) those of clean wood waste (Q1+Q2). PAHs were found in the highest concentrations in Q3, due likely to the organic resins used in fibreboards (which are classified as Q3). This is remarkable when considering the persistency of PAHs and the large presence of Q3 in the sampled wood waste. This trend suggests that concentrations of chemicals in recycled particleboard could decrease dramatically if wood waste of classes Q3 and Q4 were excluded from the feedstock for recycling. Improvements may entail the separate collection of Q3 wood waste from clean wood waste for recycling, while Q4 wood waste is collected separately at recycling centres, and measures should focus on avoiding its mis-sorting to clean wood waste for recycling.

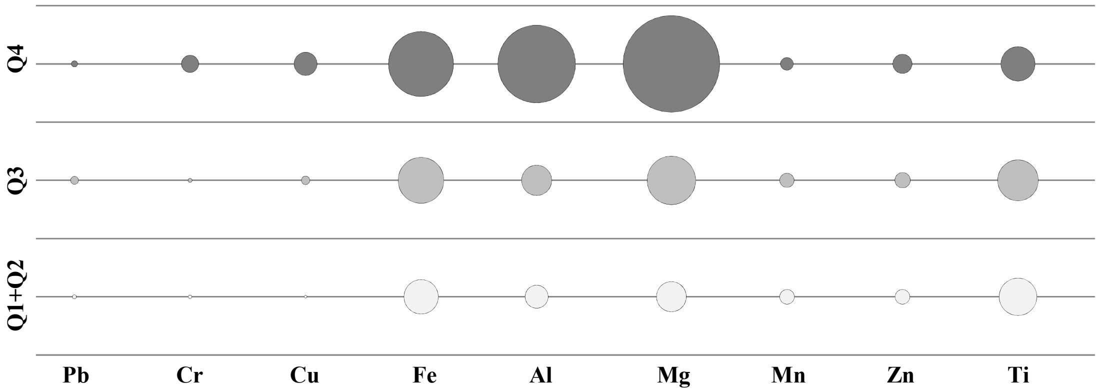

Figure 4. Concentration levels of selected chemicals in wood waste samples grouped by quality classes (cfr. Table 2). The size of the bubbles is proportional to the mean concentration of the compound. The areas of all bubbles are scaled further relative to the highest concentration in the figure (20 ppm). Adapted from Faraca et al. (**I**).

3.3 Environmental assessment of wood waste management

Based on the results discussed in Section 3.2, i.e. that selecting only the upper qualities of post-consumer wood waste for recycling would decrease the contamination level in recycled particleboard, a dynamic LCA was performed with the aim of evaluating the environmental savings that the separate collection of wood waste – according to its quality grades – would provide compared to the baseline case of collecting mixed qualities. The assessment investigated the sequential utilisation of wood waste according to a resource cascading concept (see Box 7). The development of the scenarios under assessment was based on the assumption that while collecting mixed wood waste qualities together (as in the baseline “WWmix” scenarios: one container for Q1+Q2+Q3, Q4 present as impurity) would permit particleboard production as the only possible recycling application, a separation of qualities (“WWquality” scenarios: one container for Q1+Q2 to recycling, one container for Q3 to incineration, improved separation of Q4 in a dedicated container) would enable a wider range of recycling possibilities, such as floor boards, wood insulation boards, wood plastic composites, and pellets (Vis et al., 2016). The recycling cascades under assessment and their lifetime are illustrated in Figure 5.

The characterised GWP results demonstrated that savings increased largely when recycling only the upper qualities of wood waste (Figure 6): the final net GWP100 savings achieved in WWquality scenarios B1-E1 were 8-58 times larger than WWmix scenarios A1-A4, depending on the scenario. Such savings were driven mainly by the substitution of primary wood products, which were

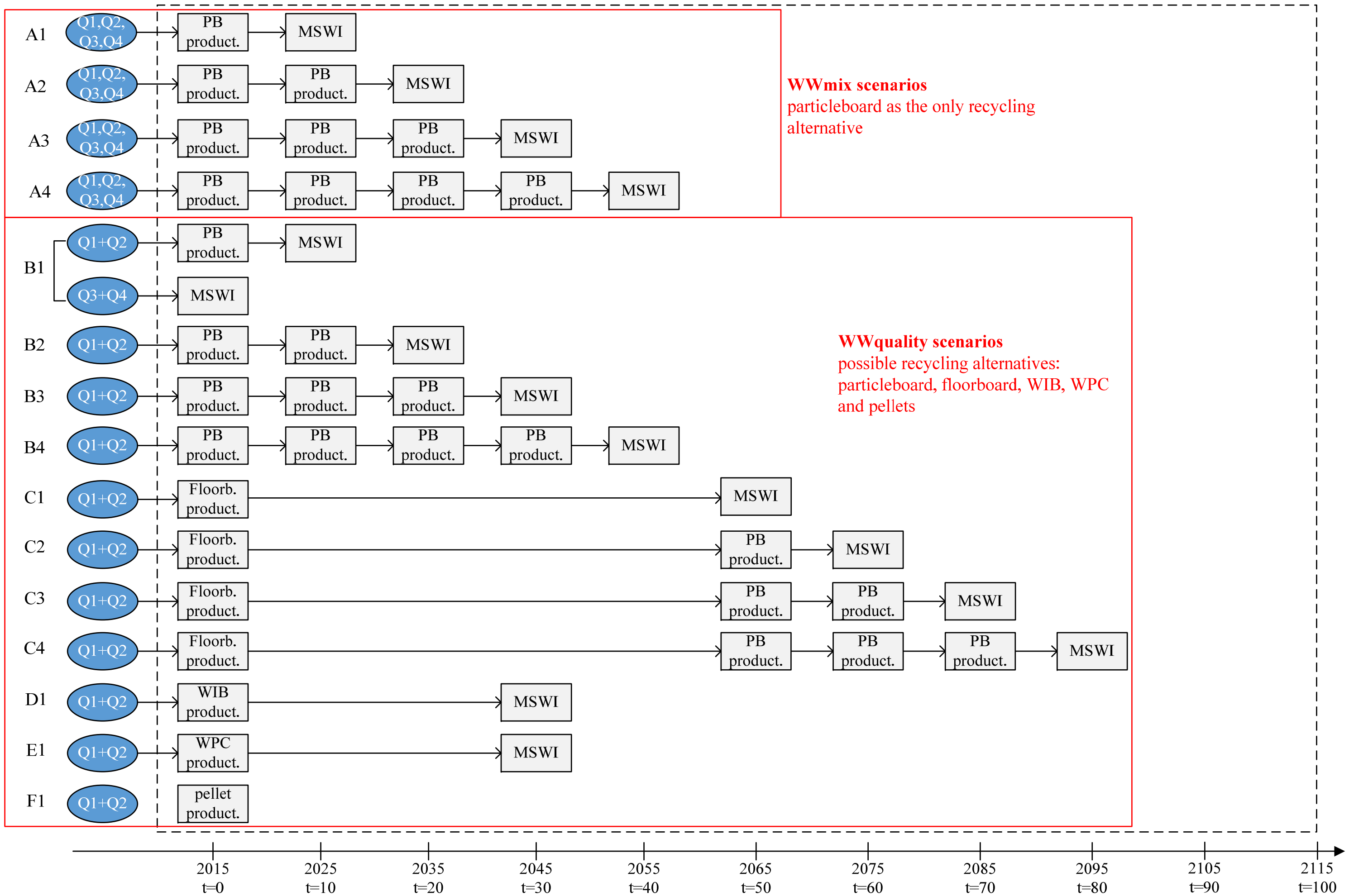

Figure 5. Description of scenarios under assessment. Floorb.=floorboard; MSWI=municipal solid waste incinerator; PB=particleboard; product. =production; WIB=wood insulation boards; WPC=wood plastic composites. In the WWquality scenarios, the management of Q3 and Q4 is maintained at a constant, as depicted exemplarily in B1: Q3 incinerated in MSWI, Q4 incinerated in hazardous waste plants. Dashed lines = system boundaries. Adapted from Faraca et al. (**II**).

larger in WWquality scenarios despite a lower mass of wood waste was sent to recycling, since Q3 wood waste was incinerated (note that biogenic carbon dioxide emissions from wood combustion were accounted for). The larger material savings in WWquality scenarios owed to a higher substitution factor (scenarios B1-B4), a longer lifetime of the recycled product (scenarios C1-C4, due to a larger carbon re-sequestration by the forest) or larger energy consumption associated with the (avoided) primary product (scenarios D1); these were all possible only when prioritising quality over quantity in wood waste management. Scenario F1 was the only management alternative resulting in a burden for the environment. This was due to the very short lifetime of the recycled pellets that ended up in combined heat and power plants at year 0, thus preventing savings from carbon re-sequestration by the re-planted forest.

Wood cascading increased GWP100 savings in both WWmix and WWquality set of scenarios (the more cascades, the larger savings), although the incremental savings provided by an additional cascade decreased for each step. Indeed, half of the wood mass was lost at each recycling step, decreasing the savings from substitution of primary materials. The first recycling step was responsible for the largest material substitution savings, proving that management choices are decisive in the earlier stages of a waste material, when resource quality is at the highest point.

Box 7. Modelling of resource cascading

Resource cascading describes the efficiency of raw material utilisation achieved through sequential utilisation. By integrating concepts of resource economics, resource cascading can be described by two dimensions, namely quality and time (Fraanje, 1997): to extend the overall usage of the resource, its properties are subsequently used in different applications as the quality declines. Resource cascading was modelled for the case of wood waste, which is feasible, as its material utilisation relies mainly on recycling to particleboard, with few niche recycling applications (floor boards, wood insulation boards, wood plastic composites, pellets) restricted by quality requirements. Due to the minor constraints on the recycling process, particleboard was assumed as the only recycling application that could absorb the second generation of recycled wood waste (cfr. Figure 5). A maximum of four cascade steps was assumed feasible, conforming to Höglmeier et al. (2014), Risse et al. (2017) and Suter et al. (2016). The modelling of subsequent recycling iterations to particleboard considered a loss of half of the waste material in input at each recycling cycle (Vis et al., 2016), reflecting the reduced quality of the feedstock material (e.g. smaller dimensions and larger presence of *Lower quality material*).

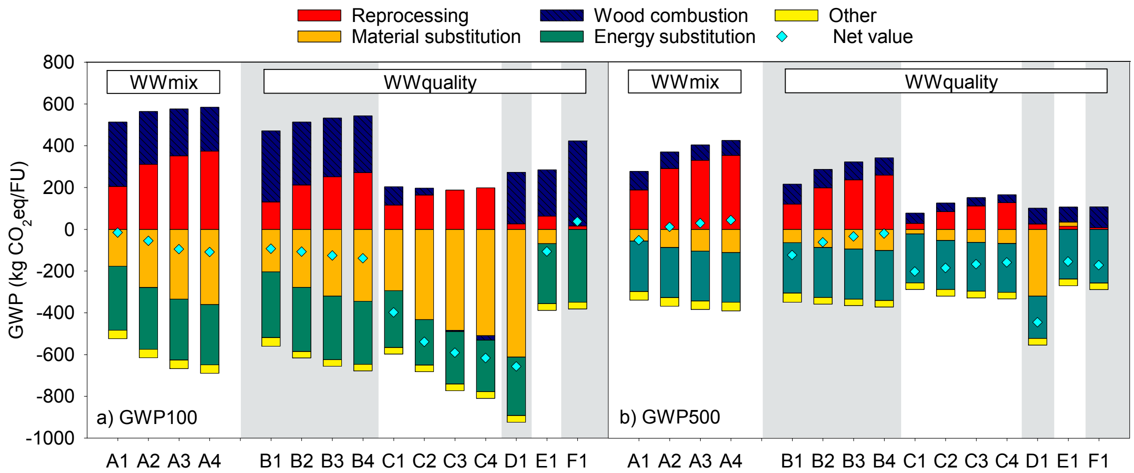

Figure 6. Global warming potential results (kg CO_2-eq/FU) for time horizons of a) 100, and b) 500 years. Only biogenic and fossil CO_2 emissions included. "Other" includes metal recycling and landfill. Cyan diamonds represent net results. Adapted from Faraca et al. (**II**).

When varying the time horizon to 500 years, conclusions over the larger savings provided by WWquality scenarios compared to WWmix scenarios held true. However, the magnitude of the results deviated considerably, and the ranking of scenarios was also altered (Figure 6b). Indeed, the cascading trend appeared inverted: increasing the number of cascade steps decreased the savings of the system, which even became burdens in scenarios A2, A3 and A4. This owed to the fact that savings by material substitution were attributed mainly to biogenic CO_2 (avoided) emissions, which account for very little on a long term horizon because they are assumed re-absorbed by new plantations. Conversely, emissions originating during the reprocessing stage were mainly of fossil origin, originating from the production of the resin needed to bind the particleboard. Large savings were achieved in scenario F1, demonstrating that energy utilisation of wood waste may lead to GWP savings, albeit only when a long time horizon (e.g. 500 years) is considered. However, recent guiding documents (IPCC, 2014; Levasseur et al., 2016) expressed concern over the large uncertainty associated with calculation assumptions in the case of long THs. GWP500 results were reported herein in order to illustrate the time dependency of biomass systems, being the 20-year case a too short reference relative to the generally longer lifetime of wood resources.

The robustness of the GWP results was tested with respect to parameter uncertainty and system assumptions. Figure 7 shows that the uncertainty on input parameters (represented by the error bars), the selection of a certain wood species and the inclusion of other greenhouse gases (GHGs) affected the results relatively little, which conversely were affected substantially by the selection of marginal electricity (in the baseline case assumed entirely from renewable sources, changed to fossil-based sources in the scenario analysis) and the inclusion of indirect land use changes (iLUC). Indeed, in these latter cases the ranking of scenarios was altered, with fossil-based marginal electricity favouring the recycling to pellet scenario (F1), and iLUC inclusion favouring the substitution of land-intensive products over energy-intensive alternatives (scenario D1). A thorough evaluation of the electricity system is thus essential to avoid misinterpreting results (the ranking of scenarios did change); similarly, an evaluation of the consequences of wood waste management systems on the global land market should be estimated and included in the system.

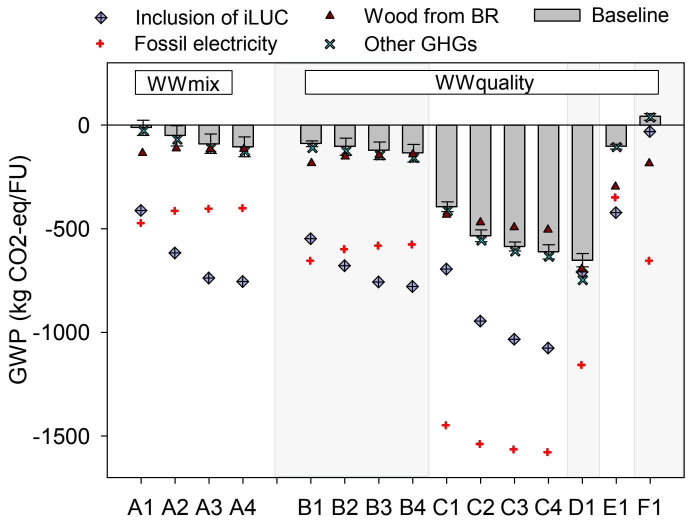

Figure 7. Global warming potential (GWP) results for TH=100 years under different assumptions. Adapted from Faraca et al. (**II**).

3.4 Recommendations for wood waste management

Sampling and characterisation activities on post-consumer wood waste collected at recycling centres and destined for recycling proved crucial for assessing the recyclability of the wood resource. The results showed that by knowing the product application of the wood waste it is possible to estimate its contamination level. For example, *C&D* may likely be contaminated by *Lower-quality materials*; *Furniture* may likely contain high levels of chemicals, e.g. PCP and PAHs; *Packaging* may likely be the cleanest wood category. Moreover, it was observed that the presence of Q3 wood waste is likely to increase the level of chemicals, even exceeding threshold limits for recycled particleboard. This suggests that the chemical composition of the wood feedstock should be monitored at recycling facilities in order to ensure safe recycling loops.

Composition results can be used to improve the collection of wood waste at recycling centres. One possibility is to separate the collection of wood waste of

Q1+Q2 grades (sent to recycling) from the Q3 grade (directed to incineration) through separate containers. The container for Q3 should include wood treated for outdoor use and fibreboards (see Table 2), which should be feasible to achieve at recycling centres, where trained staff can help the citizens to sort correctly the waste. Further investigations should identify other wood products characterised by large presence of chemicals, in order to ensure their exclusion from the Q1+Q2 container.

Separate collection according to quality grades demonstrated increased environmental savings in terms of global warming potential. The upper qualities of wood waste should preferably be recycled to floorboards or insulation boards, which can ensure substantial savings due to the substitution of long-lived or energy-intensive product. Such products may also potentially provide larger savings in other impact categories compared to particleboard. In fact, the manufacture of resin for use in particleboard was shown to have large impacts in many impact categories (e.g. Hoeglemeier et al., 2014). In addition, a separate collection may potentially mitigate the arising competition of uses between the energy and material sectors, since part of the wood waste generation would be destined for incineration. Despite the results appeared to depend considerably on certain background conditions (e.g. electricity mix), conclusions on the largest GWP savings achieved through the separate collection of wood waste were always confirmed.

The methodology adopted by the study highlighted that the assessment method in LCA should reflect transparently all variables occurring in the system. Biogenic and fossil CO_2 emissions generally account for the majority of GHG emissions in biomass systems (as also illustrated in Figure 7). Although commonly considered climate-neutral, biogenic CO_2 emissions should be included in order to provide the mass balance of carbon flows in the system. Similarly, a dynamic approach to GWP should be taken whenever emissions occur at a different point in time (e.g. anticipated or delayed) in the life cycles of the system.

4 Plastic waste

This section addresses the results pertinent to the characterisation of plastic waste from recycling centres and intended for recycling, linking it to the potential sorting efficiencies and related environmental and financial benefits obtainable from improved recycling processes.

4.1 Plastic waste material composition

Plastic waste showed a diversified product application composition between hard plastic, plastic film and PVC waste. The main product application categories in hard plastic waste were *Non-food packaging* (34% ±2%), *Food packaging* (14% ±6%), *Other* (13% ±7%) and *Construction* (14% ±4%; Figure 8a). Concerning plastic films, *Non-food packaging* alone accounted for 80% of the waste, while PVC comprised exclusively products used in *Construction*. Given that available studies on the composition of plastic waste from households reported the share of non-packaging applications in the 10-26% range (Eriksen et al., 2018; Feil et al., 2016), plastic waste from recycling centres can be considered fundamentally different, thereby calling for specific recycling measures. Moreover, odd-sized plastic products (one dimension larger than one metre), absent in plastic waste from households, represented around half of hard plastic and plastic film waste and 80% of PVC waste from recycling centres.

The quality of plastic waste was less diversified than product composition (Figure 8a): *Low quality* plastic comprised the entirety of plastic film and PVC waste, and it also represented the majority of hard plastic waste (73%), with *High quality* and *Medium quality* applications accounting for 14% and 8%, respectively. This composition anticipates that recycling may be a complex matter for the majority of plastic waste.

On average 5-8% of hard plastic and plastic film waste was made up of non-plastic products (*Misplacements*, Figure 8a). Moreover, 8% of hard plastic products were *Non-plastic interfering materials*, i.e. non-plastic parts attached to the plastic body (Figure 8b). Their presence was larger in some of the application categories, such as *Electrical and electronics* and *Other* (containing 29% and 18% of interfering materials, respectively) compared to *Food packaging* (2%), therefore suggesting that recycling may be more complicated for some applications than for others. An additional 0-8% and 3-10% of plastic waste comprised *Multi-polymer* and *Co-polymer* products, respectively. These types of impurities may affect recycling by lowering process efficiencies and causing contamination from unsorted products. For example, in the case of

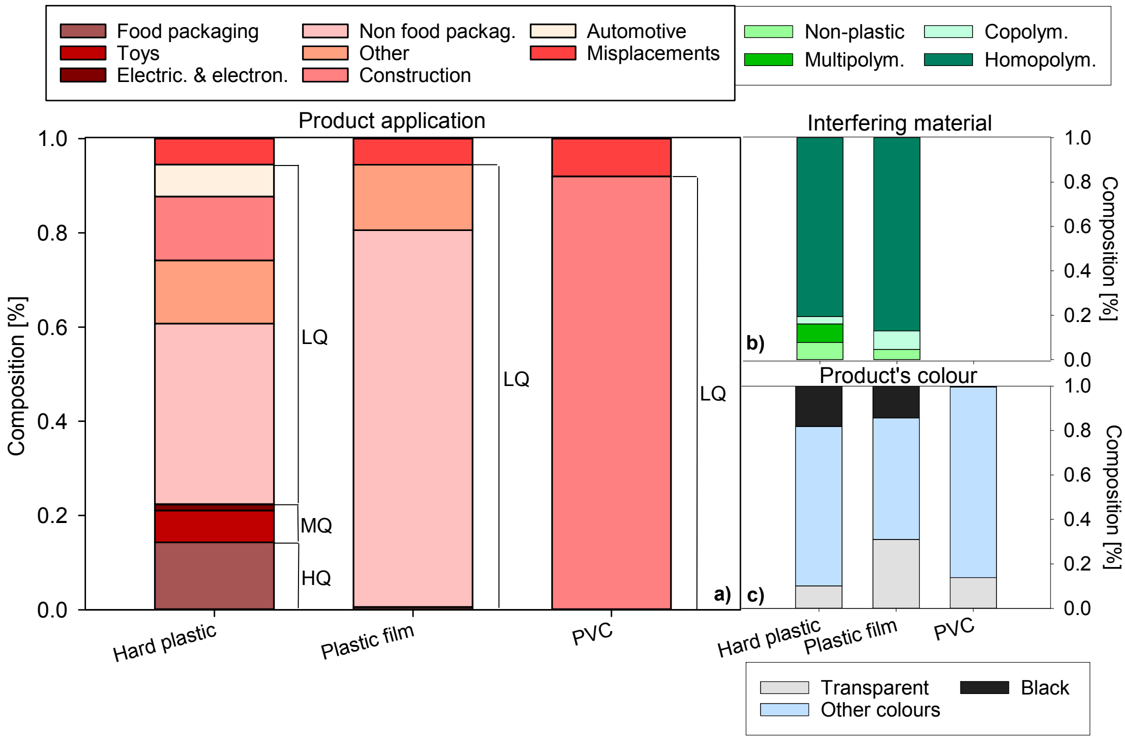

Figure 8. Composition (% wet weight) of hard plastic waste, plastic film waste and PVC waste in terms of a) product applications, b) presence of interfering materials and c) colour of the products (average over three recycling centres). In b) PVC was not investigated. HQ: High quality; MQ: Medium quality; LQ: Low quality. Adapted from Faraca and Astrup (**III**).

optical sorting, the polymer separation of multi-polymer products will depend on the side of the product facing the sensor, causing the contamination of one of the polymers.

The majority of plastic waste fractions consisted of coloured products (56-86%, Figure 8c). Black plastics accounted for 14-18% of hard plastic and plastic film waste, but they were negligible in PVC waste. This may represent a noteworthy issue, as coloured and black plastics contain pigments which may cause the chemical contamination of recycled products, on top of lowering the aesthetics of the recyclates. Moreover, black plastics cannot be recognised via infra-red optical sorting, since the carbon black pigment absorbs the light wavelength (Dvorak et al., 2011). Only 10% of hard plastic was transparent plastic, as well as 30% of plastic film and 14% of PVC waste.

The polymer composition of plastic waste is crucial for designing sorting and recycling facilities, in terms of number and type of sorting steps, expected contamination (additives, stabilisers, plasticisers, etc.) and which polymers will be targeted for recycling (reprocessing small amounts of a polymer may be uneconomical; Ardente and Mathieux, 2012). The polymer composition of plastic waste from recycling centres varied considerably in line with the plastic waste fraction (Figure 9). Hard plastic comprised mainly PP, HDPE and engineered polymers, together making around 80% of hard plastic waste. Conversely, the presence of LDPE and PET was minor, despite being considered two of the major polymers in European plastic waste from households (Delgado et al., 2007; Villanueva and Eder, 2014). Since PET is used mostly in food packaging applications (Faraca et al., **III**; Plastics Europe, 2018), its absence in plastics waste from recycling centres may be due to the high rate of food packaging products collected separately at households. On the other hand, the absence of LDPE probably owed to the separate collection of plastic films in place at recycling centres, where LDPE made up 63% of this plastic waste fraction (Figure 9b). Although LDPE polymer is highly valuable, plastic films are unwanted when recycling hard plastic, because trapping in the machineries, causing damage to equipment as well as material losses (Horodytska et al., 2018). This suggests that the separate collection of plastic film waste could ensure that films do not contaminate hard plastic streams but can be handled separately, thus potentially increasing plastic films recovery rates.

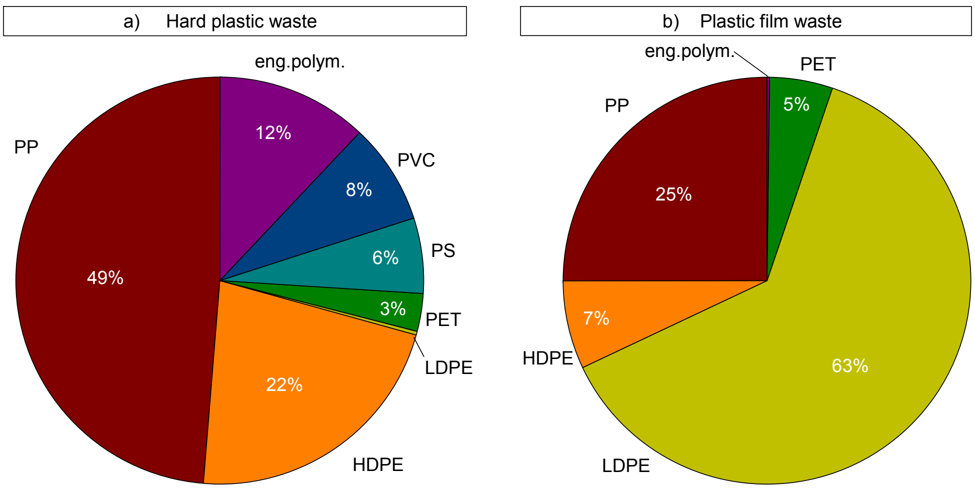

Figure 9. Polymer composition (% wet weight) of a) hard plastic and b) plastic film waste (average over three recycling centres). eng. polym.: engineered polymers. Adapted from Faraca et al. (**III**).

4.2 Recyclability of plastic waste from recycling centres

Once the levels and types of impurities in plastic waste have been identified, these characteristics can be used in MFAs to determine *a priori* the efficiencies of the system, given the technology used. Since the recycling of Danish plastic waste currently takes place abroad, hard plastic, plastic film and PVC waste flows were assumed to be treated in typical European recycling facilities. Since the number and sequence of process steps may vary across existing facilities, and in the absence of precise information, the most common recycling configurations were selected from available publications. Recycling systems were assumed to comprise a sorting step (to obtain clean homopolymer, i.e. single polymer, streams) and a number of reprocessing lines (one for each homopolymer) to produce secondary plastic materials. While sorting is generally achieved by optical sorting via NIR sensors, reprocessing technologies may have several declinations (e.g. Horodytska et al., 2018; Ignatyev et al., 2014; Kaysen et al., 2015; Ragaert et al., 2017, Villanueva and Eder, 2014).

The recycling configuration of hard plastic waste was assumed to include a NIR separation step targeting PP, HDPE, PVC, PS and PET polymers. Homopolymer reprocessing included grinding, density separation, magnetic and eddy current separation, washing, drying and extrusion to pellets. The recycling configuration of plastic film waste was assumed based on a NIR-sorting step targeting the polyolefins (PP, HDPE and LDPE), prior to a commingled reprocessing line comprising washing, wet shredding, drying and granulation into agglomerates. Finally, the PVC was assumed manually sorted into soft and hard PVC, with the latter only targeted for reprocessing (washing, density separation, drying and grinding to fine powder). Since the colour of the products and the presence of misplacements and interfering materials may potentially hamper recycling (Section 4.1), these (discriminant) factors can be assigned specific mass transfer coefficients (TC; cfr. Table 3). By knowing their proportions in the different qualities of the plastic products (*High*, *Medium* and *Low quality*, cfr. Table 2), the role of quality on plastic recyclability can be investigated.

The sorting and reprocessing efficiencies of plastic recycling systems varied considerably across hard plastic, plastic film and PVC waste flows (Table 5). Hard plastic waste was characterised by the lowest sorting and reprocessing efficiencies (52% overall), owing to the very complex mixture of input products. Individual efficiencies achieved by the polymers were mostly in the 72-94% range in the sorting and reprocessing phases, reflecting the waste composition (e.g. PS, used largely in *Automotive* applications, contained a significant amount of black products, thereby reducing its recovery; Faraca et al., **III**). Despite process efficiencies being calculated *a priori*, the outcomes were in agreement with the available literature. For example, Cimpan et al. (2015) reported sorting efficiencies at 70-90% in a plastic packaging recovery facility in the UK, while Cimpan et al. (2016) documented an overall recovery efficiency of 54% for sorting packaging plastic in a generic European material recovery facility.

Table 5. Sorting, reprocessing and recycling efficiencies achieved by different qualities of hard plastic, plastic film and PVC waste. Adapted from Faraca et al. (**III**).

Polymer	Sorting efficiency	Reprocessing efficiency	Recycling efficiency
Hard plastic waste			
High quality	83%	93%	77%
Medium quality	82%	76%	63%
Low quality	72%	70%	50%
Overall	**69%**	**75%**	**52%**
Plastic film waste			
High quality	87%	77%	66%
Medium quality	-	-	-
Low quality	86%	72%	62%
Overall	**83%**	**71%**	**59%**
PVC-container waste			
High quality	-	-	-
Medium quality	-	-	-
Low quality	99%	96%	95%
Overall	**82%**	**96%**	**79%**

Plastic film waste recycling appeared more efficient than hard plastic waste. Indeed, in the case of plastic films multiple polymers were targeted for commingled recycling, thereby decreasing sorting losses. However, the limited mechanical properties that can be achieved in the case of commingled recycling (a certain share of impurities is also included) may prevent the utilisation of the recycled agglomerates in pure plastic applications, thereby lowering their potential for substituting primary resources (not captured by recycling rates). PVC recovery was characterised by the highest recovery, thanks to the homogeneity of the feedstock, which kept material losses at minimum levels.

High quality plastic was characterised by the highest sorting and reprocessing efficiencies, which decreased significantly for *Medium* and *Low quality* products. This is due to the fact that the product applications associated with these qualities (cfr. Table 2) are generally associated with a significant amount of impurities (black products, non-plastic parts, multi-polymers), on top of employing a significant share of engineered polymers to resist increased stress conditions expected for these applications (engineered polymers were not targeted by the recycling configuration assumed herein). *Low quality* applications were characterised by lower efficiencies also in the case of plastic film waste. As Low quality applications represented the majority of plastic waste (cfr. Figure 8), their specific composition should be taken into account at recycling facilities. Furthermore, future focus should simplify the design of the Low quality applications products in order to minimise the presence of impurities in plastic waste. As High quality applications only included Food Packaging products, which are collected mainly from households (e.g. Cimpan et al., 2015; 106; Rigamonti et al., 2014), this plastic category could be removed from the collection at recycling centres and allowed only at households. This would imply different recycling technologies for the two waste streams, given the diverse composition and issues posed to recycling.

4.3 Environmental life cycle costing of hard plastic waste

The presence of impurities in plastic waste was investigated further in terms of potential consequences relating to the environmental and financial gains potentially obtained by recycling practices. As the type of technology affects the level of decoupling these contaminants from the targeted waste stream, additional recycling configurations were selected for further investigation. The focus was on hard plastic waste only, given the heterogeneity of the mix in terms of polymers, contaminants, applications and quality. Three archetypes of recycling configurations were identified (among the ones reviewed in Section 4.2), based on their different approaches to the presence of impurities: two mechanical recycling and one feedstock recycling configurations (see Box 8).

Box 8. Mechanical vs feedstock plastic recycling

Plastic recycling can be classified into mechanical and feedstock (or chemical) recycling according to the technology used and the output produced: (ASTM, 2000). Mechanical recycling involves a series of sorting and reprocessing steps until the waste products are mechanically transformed into pellets, flakes or granulates for use in new plastic products (Ignatyev et al., 2014). Feedstock recycling involves braking down the polymer chains of the plastic into smaller molecules (Brems et al, 2012). Technologies such as methanolysis and glycolysis produce monomers that can be used to produce new plastics, but need a homopolymer waste stream in input (for example, glycolysis is performed on PET only). Other technologies like hydrolysis and pyrolysis produce liquids that can be purified and used in the production of new plastics, synthetic fibres, lubricants, fuels or other chemical products (Yu et al., 2016). As the Waste Framework Directive defines as recycling only those processes producing secondary raw materials, feedstock recycling technologies are not considered recycling if the produced oils are used as fuel (EC, 2008).

The environmental and financial impacts associated with each recycling alternative were evaluated by Environmental LCC methodology (Section 2.5).

The three assessed recycling scenarios are illustrated in Figure 10. The simple mechanical recycling (sMR) scenario employs a coarse, optical-based sorting step. Given the absence of a shredding step, most of the interfering materials are not separated and enter the reprocessing line. The advanced mechanical recycling (aMR) scenario focuses more on the sorting phase, employing a sequence of shredding steps (which increase the degree of liberation of interfering materials) and the work of hand-pickers, in order to remove most of the impurities and separate black-coloured products. Finally, in the feedstock recycling (FR) scenario, the sorting step aims at separating PVC and shredding the plastic mix, which is reprocessed by pyrolysis (the plastic waste is heated under oxygen deficit until conversion into oils and gases). Separation of impurities is not necessary, as non-plastics will end up as residues in the char (Al-Salem et al., 2014).

The assessment results (Figure 11, more details in Faraca et al., **IV**) showed that scenario aMR provided significantly larger net savings in most impact categories, including total costs. Scenario sMR ranked second for the majority of impact categories, except for GWP and human toxicity - cancer effects, where it was by far the worst performing scenario. Scenario FR ranked last in almost all impact categories, with a relative difference of one or two orders of magnitude, except for GWP, where it ranked second, and ionising radiation potential, where it was the only scenario providing (large) savings.

Scenario aMR was the single recycling option providing savings in terms of GWP. These were mainly caused by the avoided production of primary plastics, which is source of large greenhouse gas emissions. Nevertheless, the overall plastic recycling efficiency in scenario aMR was 67%, indicating that opportunities for further enhancement remain. Scenario sMR led to considerable GWP impacts because of two main reasons. First, the presence of a single sorting technology handling a very heterogeneous mix lowered the purity of the targeted polymer streams (the sorted streams contained 10- 25% of untargeted polymers). Consequently, the substitution potential of recycled pellets decreased to take into account that unsorted components will degrade during extrusion, interrupting the otherwise continuous matrix of the recyclates (Ragaert et al., 2017). The second driver of GWP impacts was the large CO_2 emissions from the incineration of rejects, since 65% of the hard plastic waste was redirected to incineration, as a consequence of low efficiencies. Scenario FR also constituted

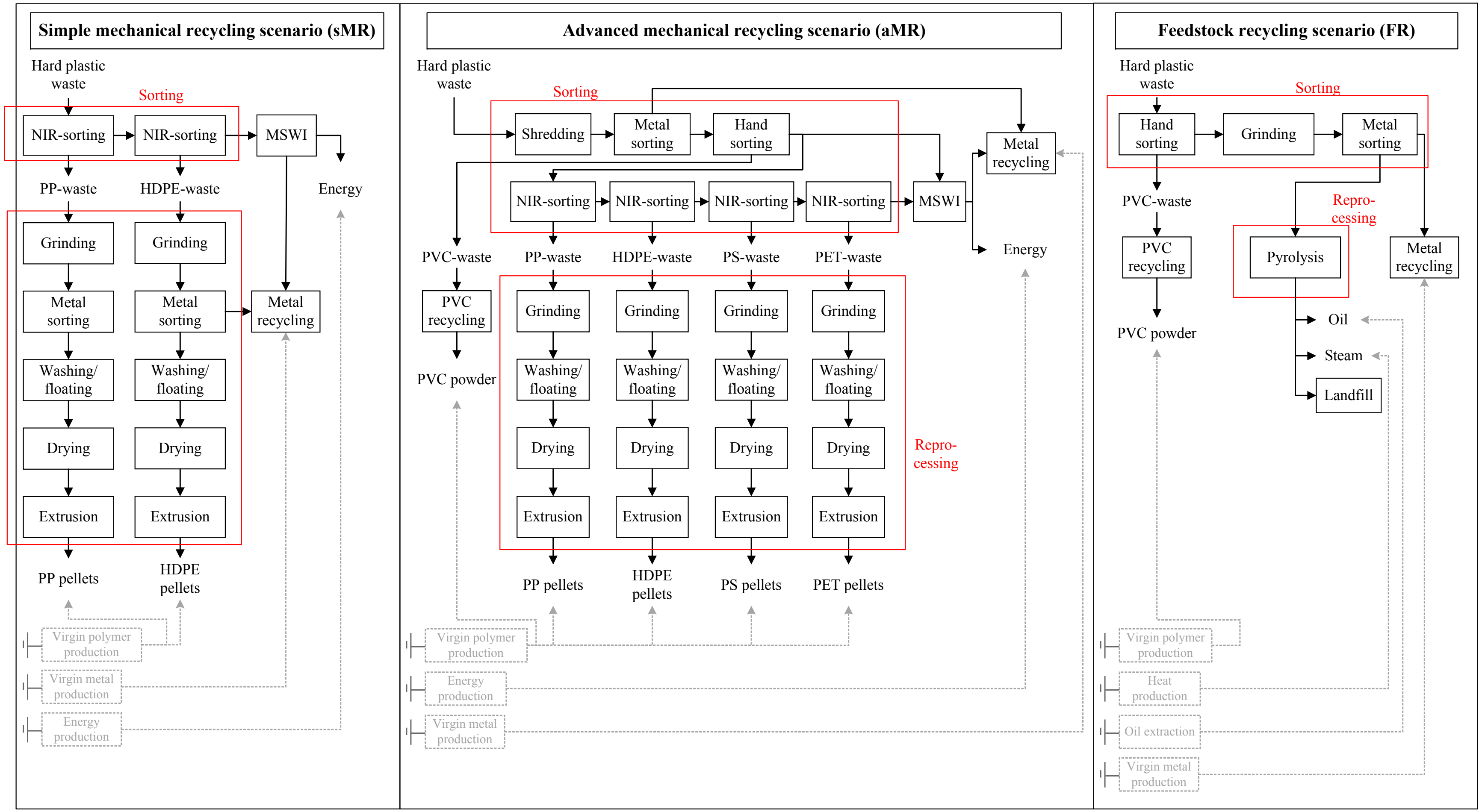

Figure 10. Description of scenarios assessed by Environmental life cycle costing. Each homopolymer reprocessing line generates losses which are sent to municipal solid waste incinerators (MSWI), although not depicted here for the sake of readability. NIR: near-infra red. Adapted from Faraca et al. (**IV**).

a burden to the environment, partly because substantial amounts of CO_2 were produced during pyrolysis, and partly because savings from substituting crude oil were not considerable. The results, while sitting within the range of the findings by other studies (e.g.Al-Saleem et al., 2014; Gear et al., 2018), support the classification of this technology as recycling only when pyrolysis oil is used in material applications (new plastics, lubricants, etc.), thereby ensuring larger savings than displayed herein.

The economic feasibility of plastic recycling was obtained only in the case of scenario aMR (-90€/FU), while scenarios FR and sMR led to net expenditures (16 €/FU and 87 €/FU, respectively). Avoided primary plastic production was the single most important factor driving financial results. In scenario aMR larger revenues were obtained not only because a larger portion of the FU was converted into secondary materials, but also because the higher polymer purity (compared to scenario sMR) was reflected by higher prices, hence outbalancing the larger budget costs required for enhanced sorting. Conversely, results for scenario sMR demonstrated that insufficient sorting may lead to a secondary material for which recycling is not economically feasible, eventually leading to failure in selling the recyclates, which in turn would nullify any potential savings from primary material substitution. Also, the low purity of the recyclates may lead to a product that cannot be used in pure plastic applications but only in filled or composite materials (Villanueva and Eder, 2014). In this case, scenario sMR would lead to a net cost of 229 €/FU (Faraca et al., **IV**). Scenario

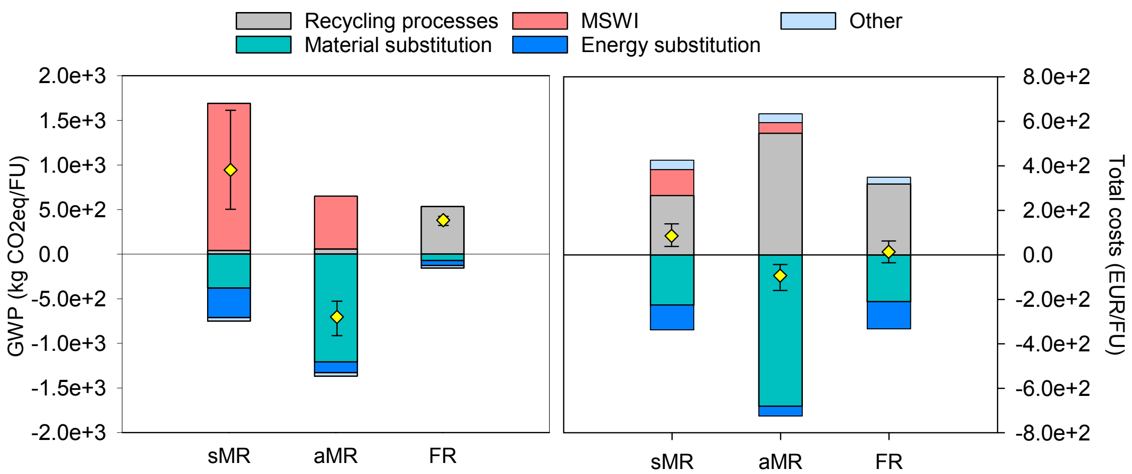

Figure 11. Characterised results for the simple mechanical recycling (sMR), advanced mechanical recycling (aMR) and feedstock recycling (FR) scenarios. FU: functional unit; GWP: global warming potential; MSWI: municipal solid waste incinerator. Yellow diamonds represent net results; error bars represent the min and max results from Monte Carlo analysis. Other includes transportation, recycling of metals and landfill of residues. Adapted from Faraca et al. (**IV**).

FR provided moderate costs, mainly driven by large investment costs for the pyrolysis plant, confirming, as advocated by Yu et al. (2016), that feedstock recycling may reach cost-effectiveness in case of large capacities involved.

Global sensitivity assessment was used as methodology to link the uncertainty of model parameters to their sensitivity on the uncertainty of the results. It was found that the variability of the results (error bars in Figure 12) was caused by three to nine parameters, depending on the impact category. The most recurrent parameters were the sorting efficiency, the reprocessing efficiency and the substitution factor; the financial results also depended on the price of (substituted) virgin plastic. This suggests thatsuch parameters should receive increased focus during modelling and when the aim is to improve the recycling system. Sections 2.1 and 2.2 showed that detailed knowledge about the plastic waste may enable the identification of sorting and reprocessing discriminant factors that can be addressed by choosing an appropriate technology.

4.4 Recommendations for plastic waste management

Plastics waste appeared to be a very heterogeneous fraction in terms of product application, quality, colour, type of polymers and impurities, especially in regard to hard plastic waste, while plastic film and PVC waste exhibited more homogeneous properties. Therefore, the current organisation of containers at recycling centres may prevent cross contamination, as plastic film and PVC waste are generally removed prior to hard plastic recycling. Moreover, it was found that the presence and type of impurities varied in line with the quality and application classes of plastic waste. For example, *Low quality* plastics were associated with larger amounts of black products and interfering materials compared to *High quality* applications. By knowing the quality/application composition of the plastics waste, the presence of factors critical to its recycling, e.g. black products, misplacements and interfering materials, can be estimated.

Detailed knowledge about the composition of plastic waste can be used in MFAs to quantify the consequences of the presence of critical materials on the overall recycling efficiency. The MFA results indicate that *Low quality* applications are characterised by larger material losses during recycling than for *High quality* applications. Since *Low quality* plastics represented more than three-quarters of the collected plastics, recycling configurations should be designed to deal with impurities. Given the substantial difference in plastic waste from recycling centres compared with streams collected at households with respect to product application and polymeric composition, separate composition-specific recycling processes could be employed to treat the waste streams, given the diversity of issues to deal with. This should be feasible given the global trade of plastics waste, enabling imports/exports of waste to be treated according to its recycling potential. Moreover, since *High quality* plastics (i.e. Food packaging) is mainly collected at households, removing this plastic category from the collection at recycling centres may help take advantage of the higher quality of plastic waste from households. Nevertheless, when the aim is to increase national recycling rates for plastics, it should be ensured that increasing the collected quantities does not affect the quality.

The presence of impurities and their effective removal were proved to be a discriminating factor for environmental savings and economic feasibility, also given the substantial share of *Low quality* plastics waste collected at recycling centres. Indeed, a recycling technology not efficiently decoupling impurities from the targeted plastic stream may lead to a large share of rejects (that need to be disposed of) and a poorer quality of the recycled plastic which may on the one hand decrease the substitutability of primary materials (reducing potential environmental savings) and on the other hand lower the final price of the recyclates (leading to cost-ineffectiveness). The characteristics that make plastics usage advantageous in many applications generate a complexity in the collected waste which hampers recycling processes. Upstream sorting holds in this case little improvement potential, as most of the contaminants cannot be sorted or even recognised by citizens. Future development should focus on simplifying the design of plastic products, especially for *Low quality* applications. When product design cannot be tailored for recycling, feedstock recycling may be a valid solution, as it eliminates the need for sorting, on the condition that the produced components are used in material applications.

5 Small combustible waste

This section addresses the characterisation of small combustible waste from recycling centres with respect to the content of mis-sorted recyclable waste and the estimation of the potential that improved collection holds for increasing national recycling rates and GWP savings.

5.1 Small combustible waste material composition

The material composition of small combustible waste (Figure 12) illustrates that *Combustible* waste accounted for 38% (median) ±15% (standard deviation) of the waste, the rest being *Non-combustible* waste (8% ±5%), *Hazardous waste* (2% ±2%) and *Recyclable* waste fractions (54% ±19%). The *Recyclable* share comprised mainly fractions such as plastics (12% ±4% of total small combustibles), textiles (12% ±6%), paper (10% ±6%) and wood (9% ±7%), while other material fractions represented a minor (cardboard, metals, WEEE and garden) or negligible (glass) contribution to the waste. Such recyclable waste materials are expected to end up in municipal incinerators along with the combustible part of the waste fraction, thereby constituting a loss of potential for increased recycling practices. In order to estimate such recycling potential, the *Recyclable* material fractions were further characterised according to properties which are relevant when addressing the recyclability of recyclable waste (cfr. Table 2). For example, plastic waste was characterised by 64% hard plastics, 37% plastic films and 8% PVC waste; hard plastic further comprised 20% High quality and 80% Medium and Low quality.

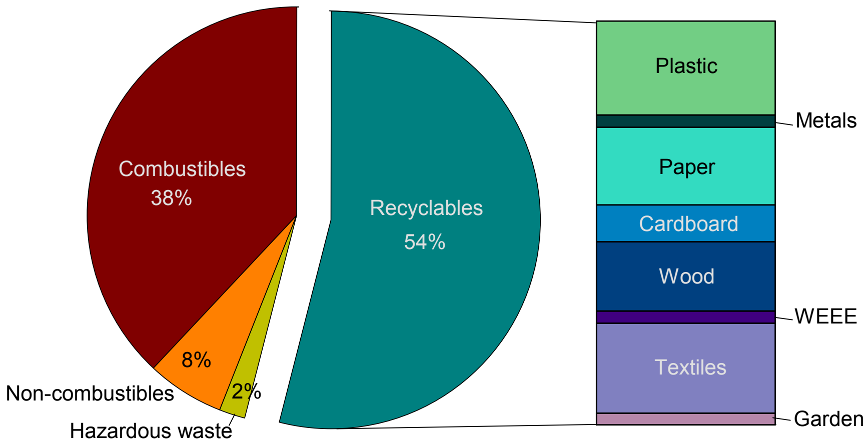

Figure 12. Pie: composition (% wet weight) of small combustible waste from recycling centres (average over eight recycling centres). Stacked bar: material composition (% wet weight) of the recyclable part of small combustible waste. Adapted from Faraca et al. (**V**).

5.2 Contribution to recycling rates

Detailed knowledge of small combustible waste composition was used to estimate the additional recycling potential that could be achieved in case the recyclable fractions included in small combustible waste were separated and sent to recycling. This was achieved by scaling up the small combustible waste composition to the quantity collected annually in Denmark and comparing it to Danish waste flows recycled in 2016 (Miljøstyrelsen, 2017c; 2018). An MFA was set and current recycling rates achieved in Denmark ("$RR_{baseline}$" indicator) were calculated and compared to the case of additionally recycling the recyclable materials in small combustible waste ("RR_{SC}" indicator). The calculation method for the recycling rate indicator (RR) follows the guidelines described in the EU (2018), which states that the RR should be calculated by considering the waste quantity entering the recycling facility, after any sorting loss (cfr. Section 2.3 and Figure 1). Textile waste not included as the primary generation data were not available for Denmark, since this fraction is often collected by charity bodies (Woolridge et al., 2006).

As illustrated in Figure 13, the additional recycling potential from small combustible waste could be relatively large for some material fractions like wood (RR_{SC} increased by 16% with respect to $RR_{baseline}$), paper (+7%) and cardboard (+6%), with plastic representing an extreme case (+47%). Conversely, glass, metals, WEEE and garden material fractions did not contribute substantially to national recycling rates (0.06%-2%), because the additional quantities contained in small combustible waste were not significant compared to national waste flows. The overall additional recycling potential provided by recyclable fractions from small combustible waste was 12% (from 26% in $RR_{baseline}$ to 29% in RR_{SC}). This denotes that the contribution of sourcing recyclable fractions from small combustible waste to overall recycling rates is not to be underestimated, particularly in view of ambitious EU recycling rates.

5.3 Contribution to environmental savings

While recycling rates are used as a mass-based metric to evaluate resource efficiency, information on the potential environmental savings provided by recycling practices is not included in recycling indicators. Therefore, the use of recycling rates was complemented with the GWP indicator. The potential GWP savings provided by correctly sorting the recyclable fractions that are currently collected as small combustibles was evaluated by LCA modelling and compared to the case of their incineration (counterfactual scenario).

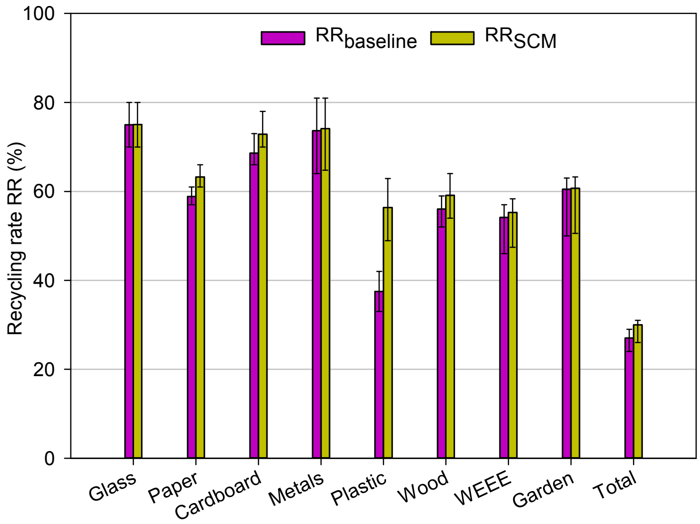

Figure 13. Recycling rate (RR) of recyclable fractions (% wet weight). $RR_{baseline}$: current Danish recycling rates (national level; Miljøstyrelsen, 2017c; 2018); RR_{SC}: recyclable fractions are separated from small combustible waste and recycled. Error bars represent the standard variation around the median. Adapted from Faraca et al. (**V**).

Figure 14 presents the GWP results provided by recycling 1 tonne of each recyclable fraction as an alternative management to incineration. Being the counterfactual alternative, the results of the incineration scenario (red-coloured bars) have been subtracted from the recycling results (green-coloured bars). The results indicated GWP savings for all recyclable fractions except for wood and garden waste, with large savings in the case of plastics, textiles, WEEE and glass. According to the processes contributing to the results, the recyclable fractions can be grouped in three classes: 1) glass, paper, cardboard and metals, for which GWP savings were driven mainly by material substitution savings in the recycling scenarios; 2) plastic, WEEE and textiles, for which the majority of GWP savings were provided by the avoided direct emissions from incineration in the incineration scenarios; and 3) wood and garden waste, for which the savings from recycling were exceeded by savings from incineration, resulting in net GWP burdens. This indicates that while for the first group of materials recycling is always beneficial, in the case of plastic, WEEE and textile waste recycling is *more* beneficial when it avoids incineration. These recyclable material fractions should be prioritised for improved sorting in order to avoid large GHG emissions.

The obtained GWP scores were highly dependent on assumptions on the electricity provision and the accounting method for biogenic CO_2 emissions, as demonstrated by the scenario analysis (Faraca et al., **V**). In particular, when

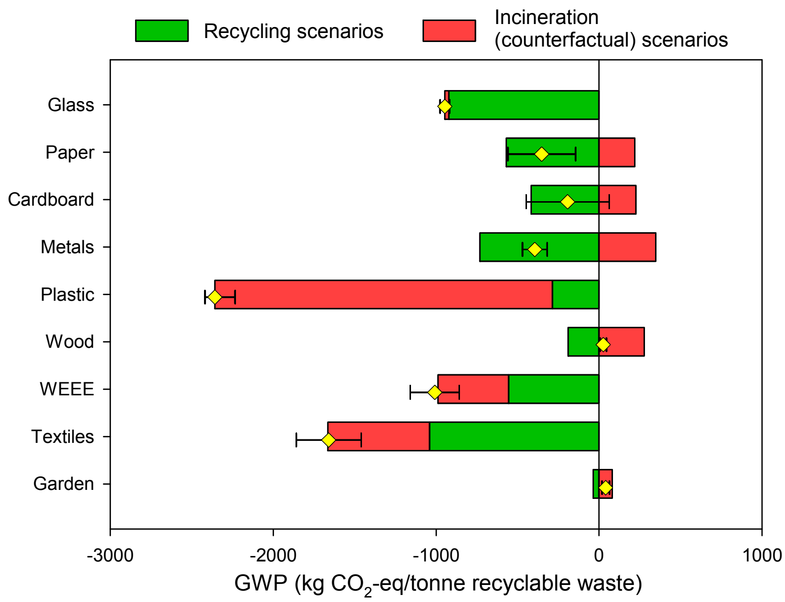

Figure 14. Global warming potential (GWP) results (kg CO_2eq/tonne recyclable waste) for the management of the nine recyclable fractions. Green bars represent the saving/impacts of recycling 1 tonne of material. Red bars represent the savings/impacts from avoided incineration of 1 tonne of the material. Yellow diamonds indicate the net results; error bars represent the min and max value from the Monte Carlo analysis. Adapted from Faraca et al. **(V).**

considering fossil-based electricity (opposed to the assumption of renewable electricity mix), great reductions in GWP savings (or increased burdens) were observed for all material fractions with high energy content. Indeed, the worsened performance was due mostly to the large savings from energy substitution in the incineration scenario, turning the recycling option less advantageous. Increased GWP savings could be observed only for glass and metal waste, since no energy is produced from their combustion, but substantial emissions are avoided when replacing primary materials. Finally, accounting for the impacts of biogenic CO_2 emissions increased the savings for all bio-based materials, even making the recycling option advantageous for wood and garden waste.

With respect to the annual quantity of recyclable waste that could be recycled in Denmark through the improved collection of small combustible waste, potential GWP savings could be 150,000 tonnes of CO_2-eq, i.e. 27 kg CO_2-eq/capita/year (GWP_{SC} bars in Figure 15, incineration scenarios subtracted from the recycling scenarios). Such savings would represent an increase of 30% compared to the current savings from recycling the same recyclable fractions ($GWP_{baseline}$, only recycling scenario, no counterfactual management assumed). This demonstrates that improvements in the collection of small combustible waste would contribute significantly to Denmark's commitment to reduced GHG emissions.

The considerable GWP savings associated with plastic and textile waste suggested that these fractions should be especially targeted for improved sorting, as they contributed to 25% of the small combustible waste and 90% of total GWP_{SC} savings. This is a noteworthy consideration, especially in light of the fact that recycling technologies for plastic and textile waste are not widely established in Europe due to a number of barriers, mainly represented by the complexity of the products. As a consequence of the recycling issues, recycling applications for plastic and textile waste are often limited to products that can deal with the low quality of the recyclates, e.g. ground surfaces in the case of textiles or plastic benches in the case of very heterogeneous plastic(as considered in the GWP results presented herein). This emphasises that GWP savings could be even larger in the near future, given the EU commitment to a circular economy. The lack of primary generation data for textile waste appears to be a key data gap to tackle, as it prohibited the comparison with the baseline scenario. However, it is estimated that 56% of textile waste is incinerated in Denmark (Schmidt et al., 2016), thereby indicating the possibly large contribution of recovering textile waste from small combustible waste.

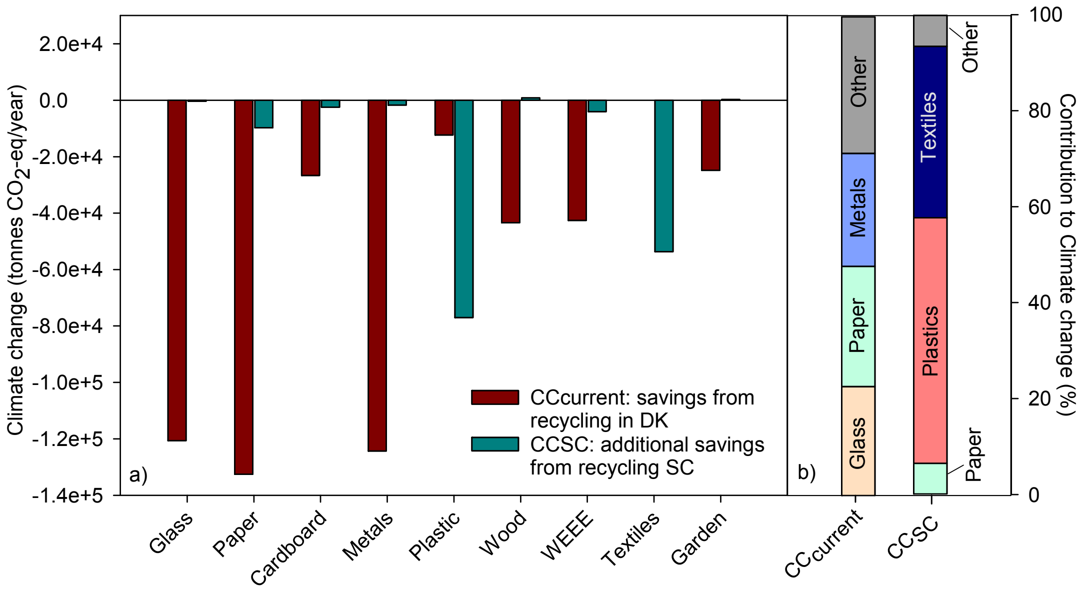

Figure 15. Global warming potential (GWP) results (tonnes CO_2-eq) yearly saved/released in Denmark from recycling individual recyclable fractions ($GWP_{baseline}$) compared to the yearly additional savings/impacts obtained by recycling recyclable fractions from small combustible and miscellaneous waste at recycling centres in Denmark (GWP_{SC}). SC: small combustible waste.

5.4 Recommendations for small combustible waste management

The results of the characterisation of small combustible waste demonstrated that the majority of this waste fraction may comprise recyclable materials, thereby constituting a significant contribution to increased recycling that is currently lost when sent to incineration. The potential consequences of improved collection/sorting, assessed by combining the use of recycling indicators and GWP results, demonstrated that large benefits could be obtained in terms of both resource efficiency and environmental impacts. In particular, plastic and textile waste should be prioritised for recycling, since these material fractions alone contributed to 25% of the small combustible waste and 90% of additional GWP savings provided by correct sorting. Collection improvements could be implemented by clear signalling at recycling centres and by special support provided by trained staff helping citizens to sort the waste. This should be feasible given the presence of dedicated containers for plastic and textile waste at recycling centres, that could be placed next to the container for small combustible waste, since Krook and Eklund (2010) advocated the effectiveness of site layout measures on the purity of collected fractions. Alternatively, a material recovery facility could be employed to separate plastic and textile waste from small combustibles. However, upstream solutions are expected to be more efficient than downstream measures, given the complex properties of plastic and textile products, which are furthermore very sensitive to degradation. Future technological development is expected to improve the effectiveness of mechanical sorting, not only increasing the quantities recovered but also widening the range of recycling applications for these material fractions (currently quite limited). Detailed knowledge of the waste material fractions was of paramount importance for thorough modelling purposes.

6 Conclusions and recommendations

This PhD thesis presented the composition of wood waste, plastic waste and small combustible waste collected at Danish recycling centres and linked it with crucial aspects in the overall recycling chains, thereby assessing the recyclability of the selected waste fractions. Moreover, the environmental and financial benefits provided by improved recycling systems were quantified. The main findings of the research can be summarised as follows:

(1) Wood waste, plastic waste and small combustible waste should not be regarded as single entities; rather, their composition should be categorised, with respect to recycling, in a number of classes (i.e. qualities) characterised by specific physico-chemical properties and levels of contamination. For example, *Construction and Demolition* wood was the most contaminated wood waste application, including 60% of material impurities; plastic waste included 25% of impurities, mostly in the form of interfering materials associated with *Non food packaging*, *Automotive*, *Construction* and *Other* applications; small combustible waste contained 54% of mis-sorted recyclable waste fractions.

(2) The recyclability of the waste fractions depends on the quality classes of the waste materials. Waste fractions including lower quality – or a mixture of qualities – limit recycling by affecting recycling efficiencies, restricting the range of possible recycling applications and reducing the potential for substituting primary materials. For example, recycling *Low quality* plastic waste would decrease system performance by 40% compared to recycling *High quality* plastic waste. Therefore, recycling treatments should be designed according to the quality class of the waste.

(3) Consistent modelling of the environmental and financial impacts of the management of wood waste, plastic waste and small combustible waste needs to reflect thoroughly the quality composition of the waste and its role in the recycling chain. In particular, environmental and financial savings in this regard are larger when isolating the highest qualities of the waste materials. In fact, global warming potential savings from recycling would increase by 8-58 times for wood waste and by two times for plastic waste when prioritising quality over quantity. Similarly, redirecting to recycling the recyclable fractions contained in small combustible waste would increase current national environmental savings from recycling by 30%.

All in all, detailed characterisation of waste material fractions was evinced necessary to identify aspects potentially hampering recycling processes, appropriately formulate alternative management scenarios and thoroughly model the associated environmental and financial impacts.

Based on the conclusions, the following recommendations are provided:

- **(1)** The collection of wood waste at recycling centres should be organised in three classes (containers): one targeting clean wood waste for recycling (within classes Q1 and Q2 as defined by the German classification for wood waste), one targeting wood waste in the form of fibreboards or wood previously utilised in outdoor applications for incineration (Q3 class), and one targeting impregnated wood waste for the disposal of hazardous waste (Q4 class, already in place at recycling centres). Such a collection measure could ensure that chemical levels in recycled products are kept at low concentrations. Q1+Q2 wood waste should preferably be used as feedstock in the production of floorboards or wood insulation boards.
- **(2)** It should be ensured that hard plastic waste from recycling centres is recycled by a mechanical treatment including an advanced sorting phase prior to reprocessing. Efforts should focus on the efficient separation of plastic polymers, especially targeting the presence of interfering materials. Minimising material losses during sorting and reprocessing would provide significant environmental savings.
- **(3)** The collection of small combustible waste for incineration purposes should be improved substantially to avoid that the recycling potential of valuable resources is lost. A combination of careful assistance and clear guidelines is needed for preventing the presence of recyclable fractions in small combustible waste at recycling centres. Plastic waste and textile waste should receive increased sorting focus.
- **(4)** Detailed sampling campaigns should be performed regularly by municipalities to strengthen the knowledge about the waste collected at recycling centres in order to improve the quality of the waste fractions and therefore increase environmental and financial benefits.

7 Future Perspectives

Based on the knowledge and experience gained during this PhD project, some objectives for future activities are suggested.

- In order to strengthen the representativeness of the characterisation results, future research may wish to repeat the sampling activities, covering a larger number of locations and time periods. This would enable the estimation of the spatial and temporal variability in the composition of the selected waste fractions.
- Wood waste cascading utilisation may lead to higher levels of chemicals in the wood product matrix as the number of cascade steps increases, potentially posing safety risks for consumers. However, exposure impacts from the presence of such chemicals cannot be captured by LCA methodology. Therefore, future research could focus on establishing a scientific framework integrating LCA and risk assessment.
- The management of wood waste may have significant impacts not only from a climate change point of view, but also with respect to the biodiversity losses incurred by wood plantations. Future research is needed to find a methodological consensus to include biodiversity aspects in the assessment of wood waste management.
- It has been demonstrated that the recyclability of plastic is affected by the presence of products characterised by a lower quality due to material contamination. Further research may want to extend the investigation to including an evaluation of the presence of chemical impurities. Indeed, additives (flame retardants, stabilisers, pigments, process aids) may represent a non-negligible share of the chemical composition of the plastic product, thereby potentially affecting reprocessing.
- Most of the impurities present in plastic waste were of the interfering materials type. As citizens' sorting efforts are expected to be insufficient for achieving a cleaner waste stream, further research may wish to investigate the environmental and financial effects of decisions made at the product design level.

9 References

Abb, M., Breuer, J. V, Zeitz, C., Lorenz, W., 2010. Chemosphere Analysis of pesticides and PCBs in waste wood and house dust. Chemosphere 81, 488–493. https://doi.org/10.1016/j.chemosphere.2010.07.042

Altholz V (2012) Verordnung über Anforderungen an die Verwertung und Beseitigung von Altholz (Altholzverordnung). (Ordinance on requirements for the recovery and disposal of waste wood (Waste Wood Ordinance). In German

Al-salem, S.M., Evangelisti, S., Lettieri, P., 2014. Life cycle assessment of alternative technologies for municipal solid waste and plastic solid waste management in the Greater London area. Chem. Eng. J. 244, 391–402. https://doi.org/10.1016/j.cej.2014.01.066

Ardente, F., Mathieux, F., 2012. Integration of resource efficiency and waste management criteria in European product policies – Second phase. https://doi.org/10.2788/72577

Ardente, F., Mathieux, F., 2014. Identification and assessment of product's measures to improve resource efficiency: the case-study of an energy using product. J. Clean. Prod. 83, 126–141. https://doi.org/10.1016/j.jclepro.2014.07.058

Arena, U., Mastellone, M.L., Perugini, F., 2003. Life Cycle assessment of a plastic packaging recycling system. Int. J. Life Cycle Assess. 8, 92–98. https://doi.org/10.1007/BF02978432

ASTM Standard D5033, 2000. Standard guide to development of ASTM standards relating to recycling and use of recycled plastics. West Conshohocken. ASTM International. https://doi.org/10.1520/D5033-00

Astrup, T., Riber, C., Pedersen, A.J., 2011. Incinerator performance : effects of changes in waste input and furnace operation on air emissions and residues. https://doi.org/10.1177/0734242X11419893

Benavides, P.T., Sun, P., Han, J., Dunn, J.B., Wang, M., 2017. Life-cycle analysis of fuels from post-use non-recycled plastics. Fuel 203, 11–22. https://doi.org/10.1016/j.fuel.2017.04.070

Bergeron, F.C., 2014. Assessment of the coherence of the Swiss waste wood management. Resour. Conserv. Recycl. 91, 62–70. https://doi.org/10.1016/j.resconrec.2014.07.011

Bilitewski, B., 2012. The Circular Economy and its Risks. Waste Manag. 32, 1–2. https://doi.org/10.1016/j.wasman.2011.10.004

Bisinella, V., Astrup, T.F., Højlund, T., 2017a. Future scenario development within life cycle assessment of waste management systems. Technical University of Denmark.

Bisinella, V., Brogaard, L. K., & Astrup, T. F. (2017b). Material flow analysis of the Danish solid waste management system. Copenhagen. Report for the Danish EPA.

Bisinella, V., Götze R., Conradsen, K., Damgaard, A., Christensen, T.H., Astrup, T.F. (2017c) Importance of waste composition for Life Cycle Assessment of waste management solutions, Journal of Cleaner Production 164, 1180-1191. http:// 10.1016/j.jclepro.2017.07.013

Boldrin, A., Christensen, T.H., 2010. Seasonal generation and composition of garden waste in Aarhus (Denmark). Waste Manag. 30, 551–557. https://doi.org/10.1016/j.wasman.2009.11.031

Bourguignon, D., 2016. Closing the loop - New circular economy package, European Parliamentary Research Service. https://doi.org/10.1111/j.1742-1241.2010.02575.x

Bouslamti, A., Irle, M.., C, B., Salvador, V., Bondu, M., Hulo, S., Caron, B., 2012. Why Simulate a Sample of Recycled Wood? Maderas. Cienc. y Tecnol. 14, 145–153. https://doi.org/10.4067/S0718-221X2012000200002

Brems, A., Baeyens, J., Dewil, R., 2012. Recycling and recovery of post-consumer plastic solid waste in a European context. Therm. Sci. 16, 669–685. https://doi.org/10.2298/TSCI120111121B

Cherubini, F., Bird, N.D., Cowie, A., Jungmeier, G., Schlamadinger, B., Woess-Gallasch, S., 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: key issues, ranges and recommendations. Res. Cons. Recycl. 53, 434-447.

Cherubini, F., Peters, G.P., Berntsen, T., Strømman, A.H., Hertwich, E., 2011. CO2emissions from biomass combustion for bioenergy: Atmospheric decay and contribution to global warming. GCB Bioenergy 3, 413–426. https://doi.org/10.1111/j.1757-1707.2011.01102.x

Cimpan, C., Maul, A., Jansen, M., Pretz, T., Wenzel, H., 2015. Central sorting and recovery of MSW recyclable materials: A review of technological state-of-the-art, cases, practice and implications for materials recycling. J. Environ. Manage. 156, 181–199. https://doi.org/10.1016/j.jenvman.2015.03.025

Cimpan, C., Maul, A., Wenzel, H., Pretz, T., 2016. Techno-economic assessment of central sorting at material recovery facilities - The case of lightweight packaging waste. J. Clean. Prod. 112, 4387–4397. https://doi.org/10.1016/j.jclepro.2015.09.011

Clavreul, J., Baumeister, H., Christensen, T.H., Damgaard, A., 2014. An environmental assessment system for environmental technologies. Environ. Model. Softw. 60, 18–30. https://doi.org/10.1016/j.envsoft.2014.06.007.

Demirbas, A., 2005. Potential applications of renewable energy sources, biomass combustion problems in boiler power systems and combustion related environmental issues. Prog. Energy Combust. Sci. 31, 171–192. https://doi.org/10.1016/j.pecs.2005.02.002

DEFRA, 2009. Municipal waste composition: A review of municipal waste component analyses. Available at: http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=15133

Dodoo, A., Gustavsson, L., Sathre, R., 2009. Carbon implications of end-of-life management of building materials. Resour. , Conserv. Recycl. 53, 276–286. https://doi.org/10.1016/j.resconrec.2008.12.007

Eriksen, M.K., Damgaard, A., Boldrin, A., Astrup, T.F., 2018a. Quality assessment and circularity potential of recovery systems for household plastic waste. Journal of Industrial Ecology. http://doi.org/10.1111/jiec.12822

Eriksen, M.K., Pivnenko, K., Olsson, M.E., Astrup, T.F., 2018b. Contamination in plastic recycling: Influence of metals on the quality of reprocessed plastic. Waste Manag. 79, 595–606. https://doi.org/10.1016/j.wasman.2018.08.007

Espinoza, L.A.T., Soulier, M., 2018. Defining regional recycling indicators for metals. Resour. Conserv. Recycl. 129, 120–128. https://doi.org/10.1016/j.resconrec.2017.10.022

European Commission (EC), 2008. Directive 2008/98/CE of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. Official Journal of the European Union.

European Commission-Joint Research Centre (EC-JRC), 2010. Institute for Environment and Sustainability: International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. EUR 24708 EN. Luxembourg. Publications Office of the European Union.

European Commission-Joint Research Centre (EC-JRC), 2011. Institute for Environment and Sustainability: International Reference Life Cycle Data System (ILCD) Handbook- Recommendations for Life Cycle Impact Assessment in the European context. EUR 24571 EN. Luxemburg. Publications Office of the European Union.

European Commission (EC), 2018a. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions – a European strategy for plastics in a circular economy, Brussels.

European Commission (EC), 2018b. Commission staff working document “Measuring progress towards circular economy in the European Union – Key indicators for a monitoring framework” accompanying the document “Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on a monitoring framework for the circular economy, Strasbourg.

European Commission (EC), 2018c. Directive (EU) 2018/852 of the European Parliament and of the Council of 30 May 2018 amending Directive 94/62/EC on packaging and packaging waste. The official journal of the European Union, Strasbourg.

European Environment Agency (EEA), 2016. Circular economy in Europe – Developing the knowledge base. EEA Report no. 2/2016. Copenhagen. Available at: https://www.eea.europa.eu/publications/circular-economy-in-europe

FAO/UNECE (Food and Agriculture Organization/United Nations Economic Commission for Europe). 2016. Forest Products. Annual market overview 2015-2016. Geneva, October 2016. ISBN: 978-92-1-117115-0

Ferreira, S., Cabral, M., Cruz, N.F., Simões, P., Marques, R.C., 2014. Life cycle assessment of a packaging waste recycling system in Portugal. Waste Manag. 34, 1725–1735. https://doi.org/10.1016/j.wasman.2014.05.007

Gear, M., Sadhukhan, J., Thorpe, R., Clift, R., Seville, J., Keast, M., 2018. A life cycle assessment data analysis toolkit for the design of novel processes – A case study for a thermal cracking process for mixed plastic waste. J. Clean. Prod. 180, 735–747. https://doi.org/10.1016/j.jclepro.2018.01.015

Gurria, P., Ronzon, T., Tamosiunas, S., Lopez, R., Garcia Condado, S., Guillen, J., ... M'Barek, R., 2017. Biomass Flows in the European Union: The Sankey Biomass Diagram – Towards a Cross-set Integration of Biomass. Joint Research Center, Seville https://doi.org/10.2760/352412.

Gy, P., 1998. Sampling for Analytical Purposes. John Wiley & Sons Ltd., Chichester, UK.

Fraanje, P.J., 1997. Cascading ofpine wood. Resour. Conserv. Recycl. 19, 21–28. https://doi.org/10.1016/S0921-3449(96)01159-7.

Fruergaard, T., Christensen, T.H., Astrup, T., 2010. Energy recovery from waste incineration: assessing the importance of district heating networks. Waste Manag. 30, 1264-1272.

Hischier R., 2007. Life Cycle Inventories of Packaging Glass. Ecoinvent report No. 11, Swiss Centre for Life Cycle Invento- ries, Dübendorf, 2007.

Hunkeler, D., Lichtenvort, K., Rebitzer, G., 2008. Environmental Life Cycle Costing. CRC Press, New York, USA.

Ignatyev, I.A., Thielemans, W., Vander Beke, B., 2014. Recycling of polymers: A review. Chem. Sus. Chem. 7, 1579–1593. https://doi.org/10.1002/cssc.201300898

Intergovernmental Panel on Climate Change (IPCC), 2013. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013. The Physical Science Basis. Working Group I Contribution to the Fifth ASSESSMENT report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA (1535 pp).

Intergovernmental Panel on Climate Change (IPCC), 2014. In: Pachauri, R.K., Meyer, L.A. (Eds.), Climate Change 2014: Synthesis Report. Contribution ofWorking Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Geneva

International Standard Organisation (ISO), 2006a. Environmental Management-Life Cycle Assessment-Principles and Framework (ISO 14040). Geneva, Switzerland.

International Standard Organisation (ISO), 2006b. Environmental Management - Life Cycle Assessment - Requirements and Guidelines (ISO 14044:2006). Geneva, Switzerland.

Jasso-Gastinel, C.F, Kenny, J.K., 2017. Modification of Polymer Properties, William Andrew Publishing, 232 pp., ISBN 9780323443531, https://doi.org/10.1016/B978-0-323-44353-1.00010-5.

Kaysen, O., Poulsen, K.G., Schmidt, A.C., Høgsted, B., 2015. Genanvendelse af hård PVC i Danmark - Miljøprojekt nr. 1717. (Recycling of rigid PVC in Denmark) Copenhagen, Denmark. In Danish

Krook, J., Mårtensson, A., Eklund, M., 2004. Metal contamination in recovered waste wood used as energy source in Sweden. Resour. Conserv. Recycl. 41, 1–14. https://doi.org/10.1016/S0921-3449(03)00100-9

Krook, J., Mårtensson, A., Eklund, M., 2006. Sources of heavy metal contamination in Swedish wood waste used for combustion. Waste Manag. 26, 158–166. https://doi.org/10.1016/j.wasman.2005.07.017

Krook, J., Mårtensson, A., Eklund, M., Libiseller, C., 2008. Swedish recovered wood waste : Linking regulation and contamination. Waste Manag. 28, 638–648. https://doi.org/10.1016/j.wasman.2007.03.010

Krook, J., Eklund, M., 2010. Developing a monitoring method facilitating continual improvements in the sorting of waste at recycling centres. Waste Manag. 30, 32–40. https://doi.org/10.1016/j.wasman.2009.08.009

Krook, J., Eklund, M., 2010. The strategic role of recycling centres for environmental performance of waste management systems. Appl. Ergon. 41, 362–367. https://doi.org/10.1016/j.apergo.2009.06.012

Lahl, U., Zeschmar-Lahl, B., 2013. Risk based management of chemicals and products in a circular economy at a global scale (risk cycle), extended producer responsibility and EU legislation. Environ. Sci. Eur. 25, 3. https://doi.org/10.1186/2190-4715-25-3

Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschild, M.Z., Christensen, T.H., 2014. Review of LCA studies of solid waste management systems - Part I: Lessons learned and perspectives. Waste Manag. 34, 573–588. https://doi.org/10.1016/j.wasman.2013.10.045

Levasseur, A., Lesage, P., Margni, M., Deschěnes, L., Samson, R., 2010. Considering time in LCA: Dynamic LCA and its application to global warming impact assessments. Environ. Sci. Technol. 44, 3169–3174. https://doi.org/10.1021/es9030003

Levasseur, A., Cavalett, O., Fuglestvedt, J.S., Gasser, T., Johansson, D.J.A., Jørgensen, S. V., Raugei, M., Reisinger, A., Schivley, G., Strømman, A., Tanaka, K., Cherubini, F., 2016. Enhancing life cycle impact assessment from climate science: Review of recent findings and recommendations for application to LCA. Ecol. Indic. 71, 163–174. https://doi.org/10.1016/j.ecolind.2016.06.049

Luijsterburg, B., Goossens, H., 2014. Assessment of plastic packaging waste: Material origin, methods, properties. Resour. Conserv. Recycl. 85, 88–97. https://doi.org/10.1016/j.resconrec.2013.10.010

Manfredi, S., Tonini, D., Christensen, T.H., 2011. Environmental assessment of different management options for individual waste fractions by means of life-cycle assessment modelling. Resour. Conserv. Recycl. 55, 995–1004. https://doi.org/10.1016/j.resconrec.2011.05.009

Mantau, U., 2015. Wood flow analysis: Quantification of resource potentials, cascades and carbon effects. Biomass and Bioenergy 79, 28–38. https://doi.org/10.1016/j.biombioe.2014.08.013

Martinez-Sanchez, V., Kromann, M.A., Astrup, T.F., 2015. Life cycle costing of waste management systems: Overview, calculation principles and case studies. Waste Manag. 36, 343–355. https://doi.org/10.1016/j.wasman.2014.10.033

Michaud, J.-C., Farrant, L., Jan, O., Kjær, B., Bakas, I., 2010. Environmental benefits of recycling - 2010 update. Report for WRAP. London. Available at: https://doi.org/10.1016/j.psym.2015.11.002

Miljøstyrelsen, 2017a. Waste statistics 2015. Environmental project nr. 1955. Available at: https://mst.dk/service/publikationer/publikationsarkiv/2018/sep/affaldsstatistik-2015/ (in Danish)

Miljøstyrelsen, 2017b. Genbrugsplast: Fra husholdningsaffald til husholdningssorteringssystem. (Recycling plastic: From household waste to household sorting system). Environmental Project No. 1951. Copenhagen, 114pp. In Danish.

Miljøstyrelsen, 2017c. Affaldsproduktion i Danmark fordelt på type og behandling (R013). (Waste producion in Denmark according to type and treatment). Available at: https://www.ads.mst.dk/Forms/Reports/ReportsBehandledeMaengderOverview.aspx (in Danish)

Miljøstyrelsen, 2018. Affaldsstatistikken 2016. Miljøprojekt nr. 2020. Available at: https://mst.dk/service/publikationer/publikationsarkiv/2018/sep/affaldsstatistik-2016/ (in Danish)

Minkkinen, P., 2004. Practical applications of sampling theory. Chemom. Intell. Lab. Syst. 74, 85–94. https://doi.org/10.1016/j.chemolab.2004.03.013

Morris, J., 1996. Recycling versus incineration: an energy conservation analysis. J. Hazard. Mater. 47, 277–193.

Muñoz, I. (2015). Example – Marginal Electricity in Denmark. Version: 2015-12-10. www.consequential-lca.org

Murphy, J., 2001. Additives for Plastics Handbook. p 484. Elsevier B.V. ISBN: 978-1-85617-370-4. https://doi.org/10.1016/B978-1-85617-370-4.50022-1

Nilsson, P., Christensen, T.H. 2011. Waste collection: Systems and organisations. Chapter 6.2 in: Christensen, T.H. 2011. Solid waste technology and management. ISBN: 978-1-405-17517-3

Pitchard, G., 1998. Plastics Additives: An A-Z Reference. Eds. Chapman & Hall, London, 633 pp., https://doi.org/10.1007/978-94-011-5862-6

Pivnenko, K., Eriksen, M.K., Martín-Fernández, J.A., Eriksson, E., Astrup, T.F., 2016a. Recycling of plastic waste: Presence of phthalates in plastics from households and industry. Waste Manag. 54, 44–52. https://doi.org/10.1016/j.wasman.2016.05.014

Pivnenko, K., Laner, D., Astrup, T.F., 2016b. Material Cycles and Chemicals: Dynamic Material Flow Analysis of Contaminants in Paper Recycling. Environ. Sci. Technol. 50, 12302–12311. https://doi.org/10.1021/acs.est.6b01791

Plastics Europe, 2018. Plastics – the facts 2017. Available at: https://www.plasticseurope.org/application/files/5715/1717/4180/Plastics_the_facts_2017_FINAL_for_website_one_page.pdf

Ragaert, K., Delva, L., Van Geem, K., 2017. Mechanical and chemical recycling of solid plastic waste. Waste Manag. 69, 24–58. https://doi.org/10.1016/j.wasman.2017.07.044

RDC Environment & Pira International, 2003. Evaluation of costs and benefits for the achievement of reuse and recycling targets for the different packaging materials in the frame of the packaging and packaging waste directive - Final Consolidated Report.

Resource Futures, 2011. Final Report to Defra: WR1002 Detailed compositional assessment for municipal residual waste and recycling streams in England. Available at: http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=17303

Rigamonti, L., Grosso, M., Møller, J., Sanchez, V.M., Magnani, S., Christensen, T.H., 2014. Environmental evaluation of plastic waste management scenarios. Resour. Conserv. Recycl. 85, 42–53. https://doi.org/10.1016/j.resconrec.2013.12.012

Rigamonti, L., Ferreira, S., Grosso, M., Marques, R.C., 2015. Economic-financial analysis of the Italian packaging waste management system from a local authority's perspective. J. Clean. Prod. 87, 533–541. https://doi.org/10.1016/j.jclepro.2014.10.069

Rigamonti, L., Niero, M., Haupt, M., Grosso, M., Judl, J., 2018. Recycling processes and quality of secondary materials: Food for thought for waste-management-oriented life cycle assessment studies. Waste Manag. 1–5. https://doi.org/10.1016/j.wasman.2018.03.001

Schmidt, A., Watson, D., Roos, S., Askham, C., Poulsen, P., 2016. Gaining benefits from discarded textiles - LCA of different treatment pathways. Copenhagen, Denmark.

Shonfield, P., 2008. LCA of Management Options for Mixed Waste Plastics. Report prepared for WRAP. London. Available at: https://doi.org/1-81105-397-0

Sirkin, T., Houten, M., 1994. The cascade chain. A theory and tool for achieving resource sustainability with applications for product design. Resour. Conserv. Recycl. 10, 213–276. https://doi.org/10.1016/0921-3449(94)90016-7

Steffen, W., Richardson, K., Rockström, J., Cornell, S., Fetzer, I., Bennett, E., … Sörlin, S. (2015). Planetary boundaries: Guiding human development on a changing planet. Science, 348(6223), 1259855. https://doi.org/10.1126/science.aaa9629

Swaeco Danmark A/S. 2015. Plastkortlægning (Plastic mapping). Report. Copenhagen, December 7. In Danish.

Swarr, T.E., Hunkeler, D., Klopffer, W., Pesonen, H.-L., Ciroth, A., Brent, A.C., Pagan, R., 2011. Environmental life cycle costing: a code of practice. Soc. Environ. Toxicol. Chem

Teuber, L., Osburg, V.S., Toporowski, W., Militz, H., Krause, A., 2016. Wood polymer composites and their contribution to cascading utilisation. J. Clean. Prod. 110, 9–15. https://doi.org/10.1016/j.jclepro.2015.04.009

The Danish Government, 2013. Denmark without waste. Available at: https://doi.org/978-87-03026-59-5

Tillman A-M (2000) Significance of decision making for LCA methodology. Environ Impact Assess Rev 20:113–123

Turner, D.A., Williams, I.D., Kemp, S., 2015. Greenhouse gas emission factors for recycling of source-segregated waste materials. Resour. Conserv. Recycl. 105, 186–197. https://doi.org/10.1016/j.resconrec.2015.10.026

Villanueva, A., Eder, P., 2014. End-of-Waste Criteria for waste plastic for conversion, JRC Technical Reports. Luxemburg. https://doi.org/10.2791/13033

Vis, M., Mantau, U., Allen, B., 2016. Study on the Optimised Cascading Use ofWood. No 394/PP/ENT/RCH/14/7689. Final Report. Brussels 2016 (337 pp).

Yu, J., Sun, L., Ma, C., Qiao, Y., Yao, H., 2016. Thermal degradation of PVC: A review. Waste Manag. 48, 300–314. https://doi.org/10.1016/j.wasman.2015.11.041

Waste & Resources Action Programme (WRAP), 2016. Household Waste Recycling Centre (HWRC) Guide. Available at: http://www.wrap.org.uk/sites/files/wrap/INH0449_HWRC_Guidance_2015_7%20FINAL.pdf

Weidema, B., 2003. Market information in life cycle assessment. Environmental Project No. 863. Danish Environmental Protection Agency, Copenhagen.

Weidema, B., Ekvall, T., Heijungs, R., 2009. Guidelines for application of deepened and broadened LCA. Project no. 037075.

Wenzel, H., Hauschild, M.Z., Alting, L., 1997. Environmental assessment of products, volume 1: methodology, tools and case studies in product development. Chapman & Hall, London.

Werner, F., Taverna, R., Hofer, P., Thürig, E., Kaufmann, E., 2010. National and global greenhouse gas dynamics of different forest management and wood use scenarios: a model-based assessment. Environ. Sci. Pol. 13 (1), 72–85.

Wilson, J., 2010. Life-cycle inventory of particleboard in terms of resources, emissions, energy and carbon. Wood Fiber Sci. 42, 107–124.

Winder, G.M., Bobar, A., 2016. Responses to stimulate substitution and cascade use of wood within a wood use system: Experience from Bavaria, Germany. Appl. Geogr. 1–10. https://doi.org/10.1016/j.apgeog.2016.09.003

Woolridge, A.C., Ward, G.D., Phillips, P.S., Collins, M., Gandy, S., 2006. Life cycle assessment for reuse / recycling of donated waste textiles compared to use of virgin material: An UK energy saving perspective 46, 94–103. https://doi.org/10.1016/j.resconrec.2005.06.006

Overgaard, K.Z., Mosgaard, M., Riisgaard, H., 2018. Capturing uncaptured values — A Danish case study on municipal preparation for reuse and recycling of waste. Resour. Conserv. Recycl. 136, 297–305. https://doi.org/10.1016/j.resconrec.2018.04.031

10 Papers

I Faraca, G., Boldrin, A., Astrup, T.F. (2018) Resource quality of wood waste: the importance of physical and chemical impurities in wood waste for recycling. *Submitted*.

II Faraca, G., Tonini, D., Astrup, T.F. (2019) Dynamic accounting of greenhouse gas emissions from cascading utilisation of wood waste. *Science of the Total Environment 651, Part 2, 2689-2700*.

III Faraca, G., Astrup, T.F. (2018) Plastic waste from recycling centres: relevance of waste characterisation data for modelling of recycling processes. *Submitted*.

IV Faraca, G., Martinez-Sanchez, V., Astrup, T.F. (2019) Environmental life cycle cost assessment: recycling of hard plastic waste collected at Danish recycling centres. *Resources, Conservation and Recycling (in press)*.

V Faraca, G., Edjabou, M.E., Boldrin, A., Astrup, T.F. (2018) Combustible waste from Danish recycling centres – Characterisation, recycling potentials and contribution to environmental savings. *Submitted*.

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